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NITROGEN RETENTION IN URBAN LAWNS AND FORESTS

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Abstract. Lawns are a dominant cover type in urban ecosystems, and there is concern about their impacts on water quality. However, recent watershed-level studies suggest that these pervious areas might be net sinks, rather than sources, for nitrogen (N) in the urban environment. A ^{15}N pulse-labeling experiment was performed on lawn and forest plots in the Baltimore (Maryland, USA) metropolitan area to test the hypothesis that lawns are a net sink for atmospheric-N deposition and to compare and contrast mechanisms of N retention in these vegetation types. A pulse of ^{15}N-NO_3^-, simulating a precipitation event, was followed through mineral soils, roots, Oi-layer/thatch, aboveground biomass, microbial biomass, inorganic N, and evolved N_2 gas over a one-year period. The ^{15}N label was undetectable in gaseous samples, but enrichment of other pools was high. Gross rates of production and consumption of NO_3^- and NH_4^+ were measured to assess differences in internal N cycling under lawns and forests. Rates of N retention were similar during the first five days of the experiment, with lawns showing higher N retention than forests after 10, 70, and 365 days. Lawns had larger pools of available NO_3^- and NH_4^+; however, gross rates of mineralization and nitrification were also higher, leading to no net differences in NO_3^- and NH_4^+ turnover times between the two systems. Levels of ^{15}N remained steady in forest mineral soils from day 70 to 365 (at 23% of applied ^{15}N), but continued to accumulate in lawn mineral soils over this same time period, increasing from 20% to 33% of applied ^{15}N. The dominant sink for N in lawn plots changed over time. Immobilization in mineral soils dominated immediately (one day) after tracer application (42% of recovered ^{15}N); plant biomass dominated the short term (10 days; 51%); thatch and mineral-soil pools together dominated the medium term (70 days; 28% and 36%, respectively); and the mineral-soil pool alone dominated long-term retention (one year; 70% of recovered ^{15}N). These findings illustrate the mechanisms whereby urban and suburban lawns under low to moderate management intensities are an important sink for atmospheric-N deposition.

Key words: atmospheric-N deposition; Baltimore (Maryland, USA) lawns and forests; development and urban watersheds; forests; lawns and ecosystem processes; mineralization; ^{15}N; nitrification; nitrogen retention; soil; turfgrass; urban.

Introduction

Urban land use is expanding rapidly in the United States. For example, between 1982 and 1997 the amount of urbanized land in the United States increased 47% compared to population growth of only 17% over this same time period (Fulton et al. 2001). The Chesapeake Bay region, where this study takes place, is expected to see an 80% increase in developed land area by 2030 if current trends continue (Goetz et al. 2004). This increase is predicted to consume 14% of forest land in the region, primarily through exurban sprawl (Goetz et al. 2004). Given this high rate of expansion, urban areas are likely to be an important contributor to ecosystem dynamics in the region; however, little is known about the basic functional properties of these systems.

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Forest conversion brings with it two major changes in landcover, namely, increased impervious surface area and the replacement of natural vegetation with lawns. The area that is covered by lawns in the United States is estimated at 163 800 km^2 (Milesi et al. 2005), and it continues to grow rapidly. In the state of Maryland, >10% of the terrestrial surface area is covered by turfgrass (based on estimates from Milesi et al. [2005]; see Plate 1). Clearly, if we are to predict the implications of urbanization it is critical that we understand the role of lawns in ecosystem processes. Unfortunately, most turfgrass studies have been done on controlled research plots and little is known about how lawns function in the context of urban and suburban ecosystems.

The increase in the area of lawns has raised concern about water pollution associated with inputs of fertilizer and pesticides for lawn establishment and maintenance (Gold et al. 1988, 1990, Morton et al. 1988, Petrovic 1990, Milesi et al. 2005). This concern is heightened in areas that already receive high levels of atmospheric nitrogen (N) deposition. However, several recent studies have suggested that urban watersheds have a high

capacity for N retention (Baker et al. 2001, Groffman et al. 2004, Wollheim et al. 2005), indicating that there are strong sinks for N within the pervious surface areas of these watersheds. Other recent research has found that lawns have dynamic soil-carbon (C) fluxes, with potential for organic-matter accumulation and N retention (Qian and Follett 2002, Kaye et al. 2005, Golubiewski 2006, Pouyat et al. 2006). Understanding the capacity of lawns to function as N sinks could be important for predicting and minimizing the impact of development on water quality.

In this study we tested the hypothesis that lawns are a significant sink for atmospheric-N deposition to urban watersheds by adding a pulse of ^{15}N and tracing its fate in lawn and forest plots in Baltimore, Maryland. We hoped to quantify and contrast the movement and conversions of N in urban lawn and forest ecosystems to evaluate the potential impacts of forest conversion on the nitrogen dynamics of developing watersheds. Our approach was to add a small pulse of ^{15}N-labeled NO_3^-, comparable to what would be added by atmospheric deposition from a small rain event, and then track tracer movement into surface organic matter, mineral soil, roots, aboveground biomass, microbial biomass, inorganic N, and evolved N_2 gas over a one-year period. This approach allowed for evaluation of the fate of N without the dramatic alteration of existing nutrient dynamics that would be caused by a larger addition (fertilizer simulation) and fostered comparison of the inherent N-retention capacities of lawns and forests.

Methods

Site description

The lawn and forest plots in this study were located in the Baltimore (Maryland, USA) metropolitan area in association with the Baltimore Ecosystem Study (BES), a component of the U.S. National Science Foundation Long Term Ecological Research (LTER) network (study description *available online*).[4]

The four lawns were located on the grounds of University of Maryland Baltimore County, but were not turfgrass research plots. Campus lawns were chosen because of their uniform management regime, similarities in land-use history (former agriculture) and plant species and their representativeness of typical urban lawns. Each lawn (or plot) contained a mixture of tall fescue (*Festuca arundinacea* spp. *L.*), fine fescue (*Festuca* spp.), and white clover (*Trifolium repens*). These lawns have been fertilized each spring at a rate of 96 kg N/ha applied in two applications approximately two weeks apart and treated with the herbicide 2,4-D once each spring at a rate of 2.4 kg/ha. Mowing has been done at 2–3 week intervals (dependent on rainfall and subsequent growth) during the spring, summer, and fall seasons at a height of approximately 10 cm. The lawns have received no irrigation, and clippings have been left in place. These lawns have been managed in this manner for at least the past 15 years.

Forest plots were located in Gwynns Falls/Leakin Park (Baltimore City) and Oregon Ridge Park (Baltimore County). These parks are notable for their size (>400 ha) and large tracts of mixed hardwood forest. Plots were dominated by oak (*Quercus* spp.) and yellow poplar (*Liriodendron tulipifera L.*) and are described in detail by Groffman et al. (2006*b*).

Atmospheric-N deposition in the Baltimore metropolitan area is estimated at 11.2 kg$\cdot$ ha$^{-1}\cdot$yr^{-1} (Groffman et al. 2004). A lawn-management survey by Law et al. (2004) found that lawn fertilizer inputs in the area range from zero to >300 kg$\cdot$ha$^{-1}\cdot$yr^{-1} with a mean (of the lawns that were fertilized) of 97.6 kg$\cdot$ha$^{-1}\cdot$yr^{-1}, which is close to the fertilization rate used on lawns in this study.

^{15}N-pulse labeling experiment

In July of 2004, labeled nitrate (99% ^{15}N-NO_3^- as KNO_3) was added at a rate of 0.3 kg N/ha to a 3 × 3 m subplot on each of the four experimental lawn and four experimental forest plots). A backpack sprayer was used to apply the ^{15}N-NO_3^- in a 0.5-cm simulated rainfall event. Nitrate applications were performed the day after a significant rainfall (when soil moisture would be relatively high) to minimize the quantity of water needed to wet the upper soil horizons and to promote even distribution of the ^{15}N-NO_3^- tracer.

Sample collection

Samples were taken just prior to ^{15}N-NO_3^- addition to establish baseline isotope ratios and then 1, 5, 10, 70, and 365 days after the addition to evaluate the short, medium, and long-term fate of the tracer. On each sampling date, two intact cores were taken from the inner 2.5 m of each 3 × 3 m subplot to measure tracer recovery in nongaseous N pools. Cores were collected to a depth of 10 cm using a 5-cm-diameter slide-hammer corer (AMS Equipment Corporation, Knoxville, Tennessee, USA). All cores were put into coolers and taken back to the laboratory for immediate processing. Six additional soil cores were taken from each plot (outside of the 3 × 3 m subplots) to quantify bulk density. Bulk density was determined by drying soil cores at 110°C for 48 h or until no change in mass was detectable.

In lawn subplots, gaseous losses of ^{15}N tracer were measured using 29-cm (inner diameter) polyvinyl chloride (PVC) cylinder chambers with gas-sampling ports (Bowden et al. 1991, Groffman et al. 2006*a*). 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. Gaseous fluxes from the forested subplots were sampled in the same manner, but with a different style of chamber. Forest chambers consisted of 15-cm-diameter PVC

[4] ⟨http://beslter.org⟩

cylinders standing 15 cm above the soil surface. These cylinders were capped at the top when sampling.

On each sampling date, two 9-mL gas samples were taken from the sealed chambers at 0 minutes, and again at 60 minutes, using fine-needle polypropylene syringes. These gas samples were transferred to evacuated vials and stored upside down and underwater in 50-mL centrifuge tubes to minimize gaseous diffusion between the samples and the atmosphere. Gas sampling was stopped after the first 10 days of the experiment as preliminary analysis suggested that we were not able to detect movement of the tracer into the atmosphere.

Sample processing

The two intact core samples taken from each subplot were processed for fine roots, Oi-layer/thatch, aboveground biomass, mineral soil organic matter (SOM), microbial biomass, and exchangeable NO_3^- and NH_4^+, usually within 24 hours of collection, but occasionally 36–48 hours were needed to finish a block of samples. Oi-layer/thatch and aboveground biomass (lawns only; see *Discussion*, below) were first removed and set aside. Next, soil cores were broken apart and sieved with all live fine roots (<2 mm) set aside. Rocks, coarse roots, earthworms, macroscopic arthropods, and large pieces of particulate organic matter were discarded. The remaining soil from the two cores was mixed to homogenize the sample. Finally, great care was taken to separate 125 grams of root-free soil for further analysis.

Aboveground biomass (lawns only) and Oi-layer/thatch were dried at 65–70°C for 48 hours and weighed. Living fine roots were vigorously rinsed with deionized water over a fine mesh screen to remove adhering soil before drying and weighing. Lastly, the 125-g sample of root-free soil was partitioned into four 30-g subsamples for analysis of (1) total soil C and N, (2) exchangeable inorganic N, (3) microbial biomass N, and (4) gravimetric moisture, by methods described in the following paragraphs.

Mineral-soil, fine-root, Oi-layer/thatch, and aboveground-biomass samples were analyzed for C and N concentrations and isotope ratios at the Cornell University Stable Isotope Laboratory in Ithaca, New York, USA. Prior to analysis, dried soil and tissue samples were ground to a fine powder in liquid nitrogen using a mortar and pestle. After thorough homogenization, a small subsample of each (10 mg for mineral soils, 6 mg for roots and Oi-layer/thatch, and 3 mg for aboveground biomass) was weighed into a 9×5 mm tin capsule, placed in a microtiter plate with individually sealable wells, and stored in a desiccator until analysis.

Exchangeable inorganic N (NO_3^-, NH_4^+) was extracted from 30 g (wet mass) of soil with 120 mL of 0.5 mol/L K_2SO_4. Samples were agitated for 60 min at 200 rpm on an orbital-shaker table and then left undisturbed for 6 h. The supernatant liquid from each sample was then collected and filtered through Whatman number 42 filter paper (pre-rinsed with 0.05 mol/L K_2SO_4 and dried for 24 h at 100°C) into clean Nalgene bottles. Liquid samples were frozen until portions could be analyzed colorimetrically for $NO_3^-$and NH_4^+ concentration or diffused onto acidified filter disks in preparation for ^{15}N determination (see *Analytical methods*, below).

Microbial-biomass N was determined using a chloroform direct-extraction technique (Brookes et al. 1985, Davidson et al. 1989). The resulting extracts were frozen until they could be digested via the alkaline persulfate oxidation method described by Cabrera and Beare (1993). Following digestion, portions of the sample were analyzed colorimetrically for inorganic-N concentration or diffused onto acidified filter discs for N-isotope ratio determination via mass spectrometry.

Soil moisture was determined gravimetrically by comparing the wet mass of a root-free soil subsample (~30 g) with its dry mass after 72 h at 100°C in a drying oven.

Gross rates of production and consumption of NO_3^- and NH_4^+

Gross rates of NO_3^- and NH_4^+ production and consumption were measured by ^{15}N isotope dilution on paired, untreated plots using the procedure described by Hart et al. (1994). Gross production was measured by adding a small amount of ^{15}N-labeled nitrogen to the product pool (NO_3^- or NH_4^+) and measuring the isotopic dilution of this pool over the course of a short incubation (30 h in this experiment). Gross consumption was measured by a decrease in the size of the product pool over the course of the incubation. Overestimation of consumption rates may occur if tracer addition stimulates consumption of the substrate pool—a fertilization effect (Hart et al. 1994). Production rates do not suffer from this same bias since the product of the process, rather than the substrate, is labeled (Hart et al. 1994).

In early October 2004, six pairs of intact soil cores were taken from each lawn and forest plot (but not from the 3×3 m pulse-labeling subplots) for determination of gross rates of microbial N cycling. Intact cores were taken to a depth of 10 cm using a 5-cm-diameter slide hammer. Coring took place when soils were relatively moist to avoid soil-wetting effects upon addition of ^{15}N-labeled NO_3^- or NH_4^+. All cores were immediately put into coolers and transported to the laboratory.

Six cores from each plot (one each from the six pairs) were injected with 10 mL of dilute ^{15}N-labeled NO_3^- or NH_4^+ (30 mg N/L) with care taken to evenly distribute the tracer throughout each core. After 30 min (to account for initial abiotic immobilization of N), one of each pair of cores was mixed and extracted with 0.5 mol/L K_2SO_4. After 30 h the second core of each pair was mixed and extracted with 0.5 mol/L K_2SO_4. This yielded three replicate measurements of gross production and consumption of NO_3^- and NH_4^+ from each plot (for calculations, see Hart et al. [1994]). A small subsample of extractant from each core was collected and analyzed colorimetrically for NO_3^- and NH_4^+

concentration. Inorganic N in the remaining extractant was diffused onto acidified filter discs in preparation for N isotope analysis (see *Analytical methods*, below). It was assumed that background NO_3^- and NH_4^+ pools had an enrichment of 0.37 atom percent ^{15}N (Hart et al. 1994). The enrichment of our inorganic-N pools with 99% ^{15}N makes this assumption reasonable, as small deviations from 0.37 atom percent in the background pools would represent only a minor error.

Analytical methods

Soil and microbial-biomass N extracts were prepared for ^{15}N analysis using the eight-day polytetrafluoroethylene (PTFE) tape diffusion method described by Stark and Hart (1996). With this method, aqueous nitrate (NO_3^-) and ammonium (NH_4^+) were converted to ammonia gas (NH_3), which was then diffused onto acidified filter-paper discs enclosed in PTFE tape. The PTFE tape prevents the filter-paper disks from coming into contact with the solution, but allows NH_3 to diffuse onto the filters. Aqueous NH_4^+ was converted to NH_3 by increasing the pH of the solution to 13 or higher. NO_3^- was first converted to NH_4^+ and then converted to NH_3, a process that was catalyzed by adding Devardas alloy to the diffusion container. A calculated blank was used to correct for potential N contamination in samples (Stark and Hart 1996). Following diffusion, filter samples were dried in a desiccator and wrapped in tin capsules for isotope analysis.

Nitrogen-isotope composition and percentage element (C and N) were determined by the Cornell University Stable Isotope Laboratory using a Finnegan Delta Plus isotope-ratio mass spectrometer (Thermo Finnigan Corporation, Bremen, Germany) plumbed to a Carlo Erba NC2500 elemental analyzer (Carlo Erba, Milan, Italy). The isotopic composition of N_2, from field-collected gas samples, was analyzed on a Europa Geo 20-20 dual-inlet isotope ratio mass spectrometer (PDZ Europa, Norwich, Cheshire, UK) retrofitted with a helium-purged autosampling chamber to minimize sample contamination during injection.

Nitrogen concentrations (^{14}N and ^{15}N) in liquid extracts were determined independently of isotope ratios to ensure accurate measurement of N pool sizes in case of incomplete recovery of N from diffused samples. While incomplete recovery has the potential to alter the isotopic composition of a sample (via physical fractionation processes), these changes are predicted to be small relative to the highly enriched samples analyzed in this study. Incomplete recoveries, however, would have a large impact on N-concentration measurements. To more accurately measure N concentrations, a small subsample of each extract was collected and run on an OI Analytical FS 3000 continuous flow analyzer (OI Analytical, College Station, Texas, USA). NH_4^+ was analyzed by reaction with phenol and hypochlorite. Nitrate was analyzed by reduction to nitrite and subsequent reaction with sulfanilamide. Concentrations were determined by continuous-flow spectrophotometric detection of the derivatized analyte as compared to a calibration curve of known concentrations.

Calculations

The percentage recovery of N tracer in each pool was measured by multiplying the atom fraction excess of ^{15}N in samples by the size of the N pool on the subplot and dividing by the total mass of ^{15}N tracer added to the subplot. Average pool sizes from each subplot (across all time intervals) were used for calculating recovery in soil and plant pools as there were no systematic variations in pool sizes over time. For microbial biomass and exchangeable inorganic N, pool sizes were calculated using the mean measured pool size on a given subplot and day (g N/g dry soil) and the mass of 0–10 cm soil on each subplot as calculated from bulk-density cores. Mineral soil organic matter (SOM) N was calculated as total mineral-soil N minus microbial-biomass N.

Recovery rates for a given pool, measured as a percentage of total ^{15}N tracer added to each subplot, were calculated as follows (where AF is the atom fraction of ^{15}N over total N in each sample):

$$\text{Percentage } ^{15}\text{N recovery} = \frac{\text{Pool size} \times (\text{Sample AF} - \text{Background AF}) \times 100}{\text{Total } ^{15}\text{N added to subplot}}.$$

This calculation presumes that changes in N-pool size ($^{14}N + {}^{15}N$) due to tracer addition are negligible. This assumption holds due to the small quantity of N applied in this study relative to the existing N-pool sizes.

Turnover of exchangeable NO_3^- and NH_4^+ was calculated using mean pool sizes for a given subplot and mean rates of production as measured by ^{15}N-pool dilution. Gross rates of production (rather than consumption) were used for calculating turnover rates because the pool dilution method has the potential to overestimate consumption (Hart et al. 1994):

$$\text{Exchangeable } NO_3^- \text{ pool turnover} = \frac{\text{Exchangeable } NO_3^- \text{ pool size}}{\text{Gross mineralization}}$$

$$\text{Exchangeable } NH_4^+ \text{ pool turnover} = \frac{\text{Exchangeable } NH_4^+ \text{ pool size}}{\text{Gross nitrification}}.$$

Ecosystem pool sizes of mineral SOM were determined by multiplying the C concentration of mineral soils by their mass (i.e., bulk density). Ecosystem pool sizes for roots, aboveground biomass, and Oi-layer/thatch were determined directly from the dry mass of samples.

Statistical analysis

Variables measured at a single time point (gross production, gross consumption, and inorganic-N pool-

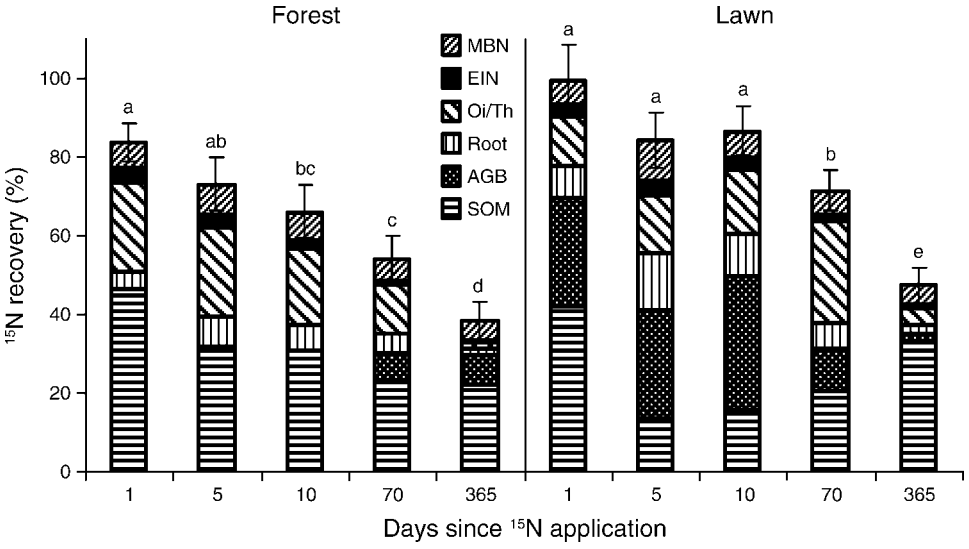

FIG. 1. ^{15}N recovery, as a percentage of applied tracer, over time for forests and lawns ($n = 4$ plots each). Full bars represent total plot recovery, which includes recovery in six ecosystem pools: microbial biomass (MBN), exchangeable inorganic nitrogen (EIN), forest Oi-layer or lawn thatch (Oi/Th), fine roots (Root), aboveground biomass (AGB), and mineral soil organic matter (SOM). Data are means ± SD. Different lowercase letters indicate significant differences in total plot recovery within and across vegetation types (at $P < 0.05$).

turnover time) were analyzed with one-way analysis of variance (ANOVA) using appropriate transformations to meet assumptions of normality. In cases where there were unequal sample sizes (due to loss of samples), a general linear model (GLM) was used. Time-series data were analyzed using repeated-measures ANOVA to test for the effects of vegetation type (lawn or forest), time, and the interaction of vegetation type and time on whole-plot ^{15}N recovery. Repeated-measures ANOVA was also used to test for these effects on the recoveries within each N pool, and to test for changes in the size of N pools (^{14}N plus ^{15}N) over time. Significant differences in recovery rates for N pools between vegetation types and days were analyzed using Tukey's honestly significant difference post-hoc tests. All statistical analyses were performed using Minitab for Windows version 14.1 (Minitab 2003).

RESULTS

Whole-plot nitrogen retention

Whole-plot recovery of ^{15}N tracer declined over time in lawns and forests (Fig. 1). Recovery in lawns declined from a high of 99% ± 9% (mean ± SD, throughout the text) one day following tracer addition to a low of 47% ± 5% after one year. Recovery in forested plots declined from 83% ± 5% to 38% ± 5% over this same time period. During the first five days of the experiment there were no statistically significant differences in total plot recovery between lawns and forests ($P = 0.18$ and $P = 0.40$ for days 1 and 5, respectively). After 10 days, lawns showed higher recoveries than forests ($P = 0.01$); however, with no estimate of aboveground biomass ^{15}N in forests for this time period (see *Discussion*, below) it is likely that forest recovery was underestimated by several percentage points. The higher N retention in lawns was particularly pronounced after 70 days, even when estimated forest aboveground biomass was included ($P = 0.01$), and this pattern continued through day 365 ($P = 0.02$). Hence, even though sample sizes were necessarily small in the present study, consistent patterns and significant main effects on ^{15}N recovery were observed.

Nitrogen recovery in ecosystem pools

Significant interactions were observed between vegetation type and time for ^{15}N recovery in fine roots, Oi-layer/thatch, and mineral-soil organic matter (SOM), indicating that N recovery differed significantly between lawns and forests over time (Table 1). Vegetation type alone was a significant predictor for these same pools. In contrast, no significant effects of vegetation type, or the interaction of vegetation type and time, were observed for the exchangeable inorganic nitrogen (EIN) and microbial-biomass nitrogen (MBN) pools. Not surprisingly, time was a significant factor in ^{15}N recovery rates for all N pools, with most pools declining over time.

Mineral SOM.—Lawns and forests showed high retention of ^{15}N in mineral SOM just one day after tracer addition with 42% ± 7% and 47% ± 4% recoveries, respectively (Fig. 2a). Following this high initial (presumably abiotic) immobilization, there were sharp drops in mineral-SOM recovery by day 5 in both lawns and forests. These drops in mineral-SOM recovery were coincident with increases in fine-root and microbial-biomass recoveries over the same time period. In forests, levels of ^{15}N recovery in mineral SOM declined

TABLE 1. Results of repeated-measures ANOVA for the effects of vegetation type (lawn vs forest), time, and vegetation type × time on recovery of ^{15}N tracer in five nitrogen pools.

Recovery of ^{15}N tracer	*P* value		
	Vegetation type	Time	Vegetation type × time
Leaf litter	0.046*	<0.001***	<0.001***
Dissolved inorganic N, DIN	0.269	<0.001***	0.686
Microbial-biomass N, MBN	0.497	0.002**	0.635
Fine roots	<0.0011***	<0.001***	<0.001***
Mineral-soil organic matter, SOM	<0.001***	<0.001***	<0.001***
Total recovery	<0.001***	<0.001***	0.069

* $P < 0.05$; ** $P < 0.01$; *** $P < 0.001$.

to 23% by day 70 and remained at that level after one year. In lawns, levels of recovery in mineral SOM continued to rise after the initial drop, up to 16% ± 4%, 20% ± 3%, and then 33% ± 4% after 10, 70, and 365 days, respectively.

Fine roots.—Lawns and forests showed similar temporal patterns of ^{15}N recovery in fine roots, but with generally higher values for the lawns (Fig. 2b). Significant amounts of ^{15}N were recovered in fine roots just one day after tracer application, with 8% ± 2.5%

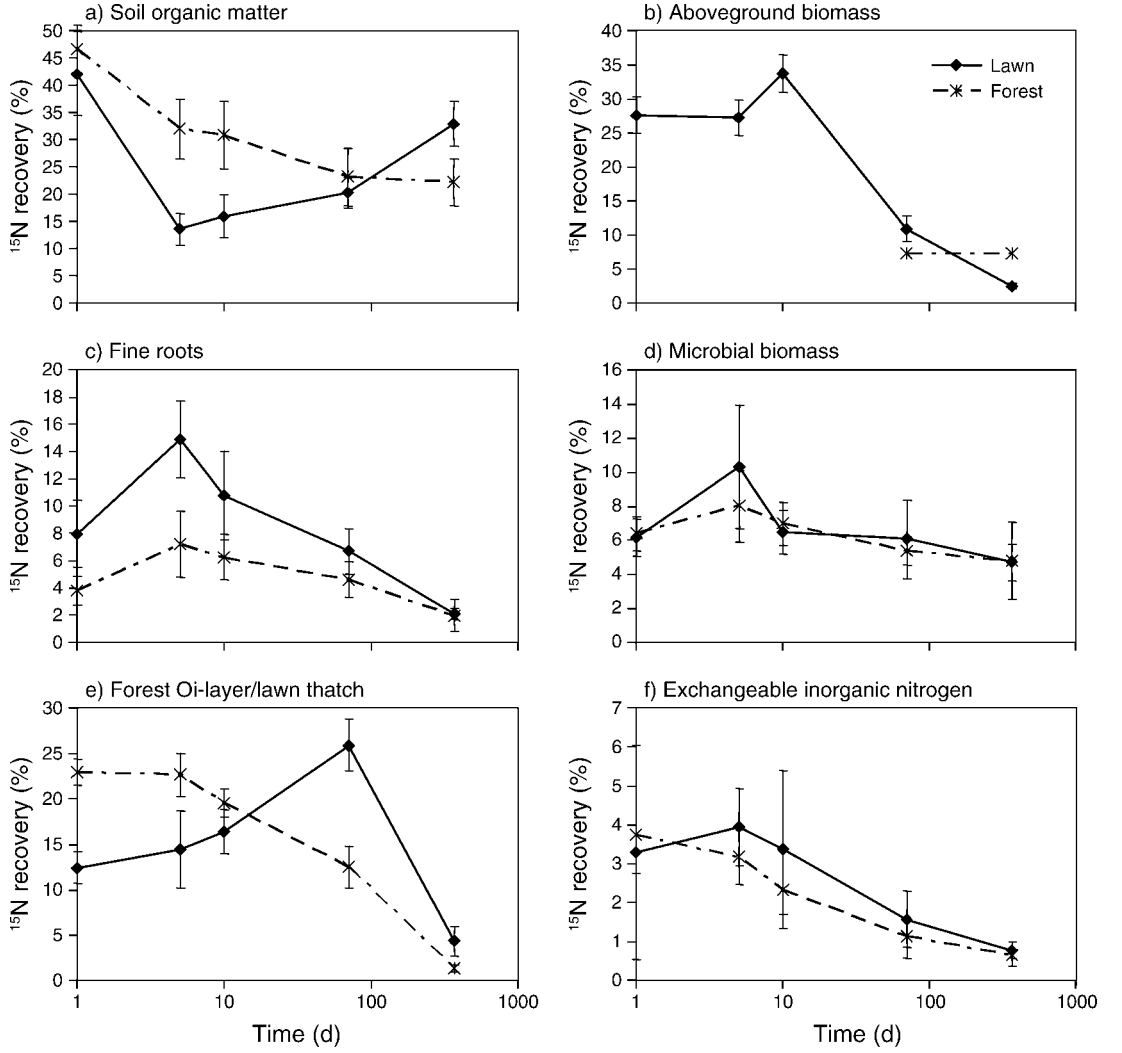

FIG. 2. Recoveries of ^{15}N in forest and lawn nitrogen pools over time for mineral soil organic matter, aboveground biomass, fine roots, microbial biomass, forest Oi-layer or lawn thatch, and exchangeable inorganic nitrogen. Data are means ± SD; $n = 4$ plots for each of forest and lawn. Note log scale of *x*-axis.

TABLE 2. Ecosystem pool sizes of organic matter and nitrogen for soil (0–10 cm), fine roots (0–10 cm), forest Oi-layer or lawn thatch, aboveground biomass (AGB), microbial biomass (MB), and dissolved inorganic nitrogen (DIN).

Variable	Plot type		Forest vs. lawn, P
	Forest (g/m^2) (mean ± SD)	Lawn (g/m^2) (mean ± SD)	
Ecosystem pool sizes of organic matter (g/m^2)			
SOM	2786 ± 184	2339 ± 89	0.014*
Root	133.5 ± 9.2	243.1 ± 28.7	0.012*
Oi-thatch	380.6 ± 8.6	311.8 ± 22.8	0.036*
AGB	n.a.	160.4 ± 55.3	
N pool size ($g\ N/m^2$)			
SOM	155.2 ± 10.3	172.1 ± 6.5	0.032*
Root	1.75 ± 0.12	2.41 ± 0.28	0.014*
Oi/thatch	4.64 ± 0.10	4.96 ± 0.36	0.489
MB	2.62 ± 1.47	3.12 ± 0.89	0.418
DIN	0.48 ± 0.16	1.07 ± 0.2	0.019*
AGB	n.a.	3.85 ± 1.33	n.a.

Note: Sample sizes were $n = 4$ plots for each site type; n.a. = not applicable (we were unable to measure tracer recovery in aboveground biomass; see *Discussion*).
* $P < 0.05$; ** $P < 0.01$; *** $P < 0.001$.

and 4% ± 1.0% of applied ^{15}N recovered in lawn and forest roots, respectively. Both vegetation types showed increases in fine-root recoveries between days 1 and 5, with 15% ± 3% of applied ^{15}N recovered in lawns and 7% ± 2.4% recovered in forests on day 5. After day 5, recoveries of ^{15}N in fine roots declined over time for both lawns and forests, with a mean fine root recovery of 2% after one year.

Oi-layer/thatch.—The pattern of ^{15}N recovery in the Oi-layer of forests and the thatch layer of lawns differed markedly (Fig. 2c). In forest plots ^{15}N recovery in the Oi-layer declined steadily, from 23% ± 1.5% after one day, down to 20% ± 1.6%, 12% ± 2.3%, and eventually 1.3% ± 0.4% after 10, 70, and 365 days, respectively. In lawns, ^{15}N recovery in thatch increased from 12% ± 2% to 26% ± 3% over the first 70 days of the experiment, presumably due to influxes of new biomass from plant growth and regular mowing. However, after one year only 4% ± 1.6% of applied ^{15}N was found in the lawn-thatch pool.

Aboveground biomass.—The ^{15}N-labelled nitrate was rapidly incorporated into aboveground biomass in lawns, accounting for 27% ± 3% of applied ^{15}N after just 24 hours (Fig. 2d). Recovery stayed relatively constant for the first 10 days of the experiment with means of 27% ± 3% and 33% ± 3% for days 5 and 10. Recoveries decreased thereafter, with means of 11% ± 2% and 2.5% ± 0.4% after 70 and 365 days, respectively.

Microbial biomass.—There were no significant differences in microbial biomass (MB) recovery between lawn and forest plots over time (Fig. 2e). Mean ^{15}N recoveries in MB fluctuated between 5% and 10%, with the highest values occurring 5 days after tracer addition (Fig. 1). Aside from that brief spike on day 5, ^{15}N recoveries remained relatively constant in the microbial pool throughout the experiment.

Exchangeable inorganic nitrogen.—Only small amounts of ^{15}N were recovered in the exchangeable inorganic nitrogen (EIN) pool, which includes both NO_3^- and NH_4^+ (Fig. 2f). Peak recoveries occurred during the first five days following tracer application (3% to 4%). The level of ^{15}N in EIN pools declined thereafter with recoveries of 0.8% ± 0.2% and 0.7% ± 0.3% for lawns and forests, respectively, after one year.

Gaseous losses.—Soil gas samples collected on days 1, 5, and 10 were not significantly enriched in ^{15}N above background samples.

Ecosystem pool sizes of organic matter and nitrogen

Soil bulk densities (measured in the top 10 cm) were higher in lawns than forests ($P = 0.01$). Mean bulk densities were 1.01 ± 0.07 g/m^3 and 0.77 ± 0.07 g/m^3 for lawns and forests, respectively. Mineral SOM pools were 2339 ± 89 g/m^2 in lawns and 2786 ± 184 g/m^2 in forests ($P = 0.01$, Table 2). Differences in fine-root biomass between lawns and forests were large, with lawns having 243 ± 29 g/m^2 and forests 134 ± 9 g/m^2 of fine roots in 0–10 cm cores ($P = 0.01$). While forest Oi-layers were significantly larger than lawn thatch, the differences (380 g/m^2 vs. 312 g/m^2, respectively) were not as large as expected. This can be attributed to the sampling dates, which fell in the summer or early fall (before tree leaf fall) of each year and to the earthworm activity in forested plots; in all cases the Oi layer was thin and consisted mostly of the previous year's leaf fall. Mean aboveground biomass in lawn plots was 160 ± 55 g/m^2. Time was not a significant predictor of gross pool sizes (based on repeated-measures ANOVA) for forests or lawns for the sampling intervals used in this study.

Nitrogen pools in mineral SOM, aboveground biomass, roots, thatch/Oi-layer, MB, and EIN, did not change significantly over time in either lawn or forest

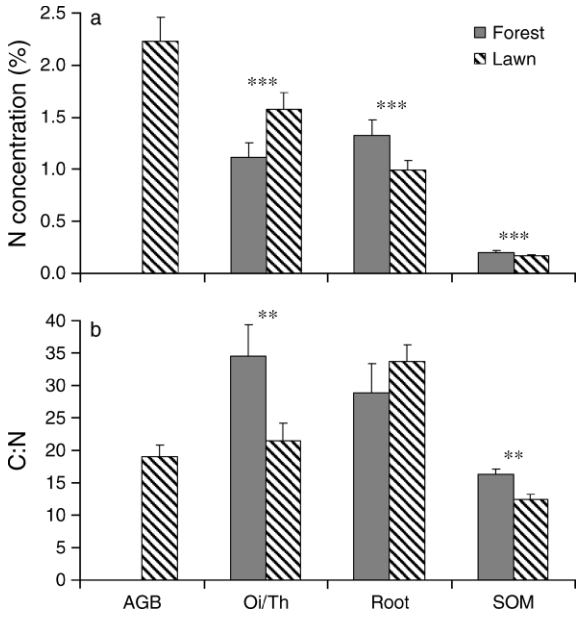

FIG. 3. Nitrogen concentration and C:N ratio of aboveground biomass (AGB), forest Oi-layer or lawn thatch (Oi/Th), fine roots (Root), and mineral-soil organic matter (SOM) in lawns and forests. Error bars represent +SD; $n = 4$ plots.
** $P < 0.01$; *** $P < 0.001$.

plots (based on repeated-measures ANOVA for the effects of time on pool size). Sampling took place in mid-summer and early fall (before leaf fall) of the first year and mid-summer of the following year so some seasonal changes in pool sizes are not represented in these time-series data, which may account for a lack of significant trends over time. Lawns had significantly larger pools of mineral SOM N (172 vs. 155 g/m^2, $P = 0.03$), fine-root N (2.41 vs. 1.75 g/m^2, $P = 0.01$), and exchangeable inorganic nitrogen (1.07 vs. 0.48 g/m^2, $P = 0.02$) than did forest plots (Table 2). Lawn and forest pools of thatch/Oi-layer N (4.96 vs. 4.64 g/m^2, $P = 0.49$) and microbial-biomass N (3.12 vs. 2.62 g/m^2, $P = 0.42$) were not significantly different.

The N concentrations in mineral soil, root, and thatch/Oi-layer differed between lawn and forest sites ($P < 0.01$ for all, Fig. 3a). Forest mineral soils had 0.20% ± 0.02% N compared to 0.17% ± 0.01% in lawn mineral soils. The N concentration in fine roots was higher in forest plots (1.33% ± 0.15%) than lawn plots (0.99% ± 0.09%). Oi-layer N concentration was lower in forests than thatch N concentration in lawns (1.11% ± 0.15% vs. 1.58% ± 0.16%). Lawn aboveground biomass contained 2.23% ± 0.23% N.

Forests and lawns differed in the C:N ratio of the thatch/Oi-layer and mineral soil ($P = 0.004$ and $P = 0.001$, respectively; Fig. 3c). The mean C:N of Oi-layer in forested plots was 34.6 ± 4.8, compared to 21.5 ± 2.7 for thatch in lawns. Mineral soil C:N ratios were also higher in forest plots (16.3 ± 0.8 vs. 12.4 ± 0.9). Mean C:N in fine roots was 28.9 (±4.5) in forested plots compared to 33.6 ± 2.7 in lawns, though this difference was not statistically significant ($P = 0.157$). The C:N of aboveground biomass in lawns was 19.07 ± 1.76.

Internal nitrogen cycling and pool turnover

Gross N mineralization and nitrification were over twice as high in lawns as in forests ($P < 0.05$; Table 3). Gross consumption of NO_3^- and NH_4^+ were >3 times as high in lawns. Despite higher rates of gross mineralization and nitrification in lawns, turnover times for exchangeable NO_3^- and NH_4^+ pools were not significantly different from forests ($P = 0.259$ and $P = 0.214$, respectively; Table 3).

DISCUSSION

The results of this study suggest that lawns have the potential to sequester atmospheric-N deposition in vegetation and surface soils at similar or higher rates than nearby forested systems do. Lawns and forests showed similarly high rates of N retention during the first

TABLE 3. Soil nitrogen-cycling parameters, including exchangeable inorganic nitrogen, gross production, gross consumption, and turnover time (0–10 cm depth); data are means ± SD.

Parameters	Plot type		Forest vs. lawn P
	Forest	Lawn	
K_2SO_4 exchangeable			
NO_3^-	0.54 ± 0.20	1.14 ± 0.32	0.020*
NH_4^+	5.63 ± 2.05	9.44 ± 1.92	0.035*
Gross production			
NO_3^-	1.44 ± 0.77	3.80 ± 1.52	0.032*
NH_4^+	2.88 ± 0.78	7.59 ± 2.13	0.002**
Gross consumption (immobilization)			
NO_3^-	1.22 ± 0.18	3.68 ± 0.55	0.002**
NH_4^+	1.77 ± 0.51	6.61 ± 1.42	0.001**
K_2SO_4 exchangeable turnover rate			
NO_3^-	0.23 ± 0.18	0.16 ± 0.04	0.259
NH_4^+	5.47 ± 3.58	3.74 ± 3.59	0.214

* $P < 0.05$; ** $P < 0.01$.

PLATE 1. The area that is covered by lawns in the continental United States is estimated at 163 800 km^2 (Milesi et al. 2005), and it continues to grow rapidly with urban development. In the state of Maryland, where this aerial photo was taken, approximately 10% of the terrestrial surface area is covered by turfgrass (based on estimates from Milesi et al. [2005]). Clearly, if we are to predict the implications of urbanization, it is critical that we understand the role of lawns in ecosystem processes. Photo credit: Aerial imagery of the Gwynns Falls watershed in Maryland, USA, by Emerge Inc., under contract with the USDA Forest Service.

five days, but after 70 days to one year lawns retained a significantly higher proportion of a $^{15}NO_3^-$ pulse than did forests. The major short-term fates of N in lawns were initial (presumably abiotic) immobilization in mineral SOM (soil organic matter) followed by rapid uptake and incorporation into plant and microbial biomass. Over the medium term, lawn thatch became an increasingly important sink for N due to regular mowing. However, at the end of one year the mineral SOM pool was the dominant sink, accounting for the majority of recovered tracer. These results suggest that the relatively rapid movement of clipping-based N into mineral SOM may be an important mechanism for long-term retention of N in urban ecosystems. High internal rates of N cycling coupled with rapid turnover of available NO_3^- suggest that N is tightly cycled in lawn and forest systems, which may contribute to high rates of retention.

The 3×3 m subplot size used in this study (chosen for reasons of logistics and cost), precluded accurate determination of aboveground uptake in the forested plots. We therefore estimated this sink for added ^{15}N for the 70-day and one-year time points based on published results from a larger-scale ^{15}N addition (30×30 m plots) done at the Harvard Forest (Massachusetts, USA) Long-Term Ecological Research site (Nadlehoffer et al. 2004). That study found recoveries of 4.69% in foliage, 2.12% in bark, and 0.46% in the two most recent years of wood (i.e., total recovery = 7.27%) on an oak-dominated hardwood plot, which received trace amounts of $^{15}NO_3^-$ in multiple additions (Nadelhoffer et al. 2004). A study by Providoli et al. (2006), which more closely resembled the format of our present experiment in that ^{15}N was added as a single addition rather than as multiple additions, found ~3–7% tracer recovery in aboveground tree biomass after eight months. That study was done in a 15-year-old plantation of *Picea abies* with dense understory vegetation, but supports the assumption that the fraction of tracer accumulation in aboveground tree biomass is <10% over these time scales.

Retention rates were high in both systems one day after the simulated atmospheric-deposition event (83% and 99% for forests and lawns, respectively) with the largest fraction of recovered ^{15}N in the mineral SOM pool. The results of previous studies suggest that some of this high initial immobilization in SOM may be abiotic (Berntson and Aber 2000, Zogg et al. 2000, Dail et al. 2001, Perakis and Hedin 2001, Providoli et al. 2006). Tracer recovery in mineral SOM declined considerably by the next sampling interval (day 5), indicating the transient nature of this initial retention. After mineral SOM, the Oi-layer SOM was the most important initial sink for N in forests with 27% of recovered ^{15}N found in this pool after one day. This high initial retention in the Oi-layer may reflect active uptake (by microbes, fungi, and plant roots) or may be

attributable in part to abiotic processes. In lawns, the thatch pool was not as large an initial sink for ^{15}N (13% recovery) as the forest Oi-layer. Instead, rapid uptake into roots and aboveground biomass accounted for the majority of non-SOM ^{15}N recovered initially (36% combined). Over the following 10 days in the lawns the N tracer was redistributed among fine roots, microbial biomass, and aboveground vegetation, and total N retention was roughly constant (Fig. 2). Although similar tracer behavior was observed in the forest, the quantity of N moving into aboveground biomass likely was much lower. The high N retention in lawn biomass probably reflected in part high grass vegetation growth during this period associated with above-normal rainfall (USGS 2007).

After 70 days we saw further redistribution of ^{15}N in lawns, including significant movement of tracer from aboveground biomass into the thatch pool, reflecting regular mowing. While thatch accounted for more than one third of total recovery after 70 days, by the end of one year only a small fraction (9%) of recovered tracer was held in this pool. The loss of ^{15}N from the thatch pool can be explained by the rapid decomposition of lawn biomass (Kopp and Guillard 2004, Shi et al. 2006). In contrast, the decline in ^{15}N recovered from the forest Oi-layer over 70 days cannot be attributed primarily to decomposition or earthworm activity because there was no significant change in the mass of the Oi-layer over this time period. The results of our experiment suggest that thatch may serve as a medium-term sink for atmospheric-N deposition before it is decomposed and incorporated into mineral SOM or lost from the system. This notion is supported by past N mass-balance studies in turfgrass systems (Miltner et al. 1996, Horgan et al. 2002*a*, Engelsjord et al. 2004), which have shown thatch to be a significant sink for fertilizer N.

In both lawn and forest systems the mineral SOM pool was the dominant long-term sink for added N, accounting for 58% of recovered tracer in forests and 70% of recovered tracer in lawns after one year. Interestingly, while the mass of tracer held in forest mineral SOM was unchanged between 70 days and one year (averaging 23% and 22% of applied ^{15}N, respectively) the mass of ^{15}N in lawn mineral SOM continued to increase throughout the experiment. There are several possible reasons for this difference between lawns and forests: (1) the lawn soils were more recently disturbed and may be aggrading N and C; (2) the higher rates of microbial immobilization in lawns may support more rapid incorporation of N into mineral SOM; (3) the greater lability (Fig. 3a) of plant tissue in lawns, combined with the regular addition of clippings, may promote the rapid movement of plant biomass into mineral SOM; and (4) relatively high rates of N fertilization combined with labile atmospheric inputs may decrease the need for plants and microbes to mine N from the more recalcitrant mineral SOM pool. Some of these possibilities are explored below.

A number of studies have shown that nitrogen and carbon concentrations in soils tend to recover following losses from soil disturbance, such as agricultural activity (see, e.g., Knops and Tilman 2000, Golubiewski 2005). A similar legacy of disturbance may be contributing to the accumulation of nitrogen tracer in SOM under these lawns, which were in agriculture before the 1960s (see *Methods: Site description*, above).

High rates of microbial immobilization and transformation of nitrogen may also play a role in the accumulation of SOM nitrogen in lawns. Gross rates of production and consumption of NO_3^- and NH_4^+ were more than double the rates seen in forest sites. High rates of microbial uptake, followed by normal cell death, may cause less labile cellular constituents to accumulate in the soil. Alternatively, a portion of the N mineralized by microbial activity may be bound to SOM via processes similar to those seen upon initial application of the ^{15}N tracer.

While mineral SOM was the dominant long-term sink, a small fraction of ^{15}N was recovered in leaf litter, plant biomass, and microbial biomass after one year, which suggests that some nitrogen continues to be cycled between the soil and plant pools. It is possible that the slow turnover of the relatively large SOM nitrogen pool limits the amount of ^{15}N cycled back into the labile pools. It is also possible that relatively large inputs of labile N to these lawns may decrease the need for plants and microbes to access more recalcitrant forms of N from mineral SOM.

NO_3^- and NH_4^+ availability, turnover, and leaching potential

Nitrogen appears to be tightly cycled in lawns and forests, with small pools of available NO_3^- and NH_4^+ and rapid turnover times, which suggests low leaching potential in both systems. Though the pools of available NO_3^- (the form of N with the greatest potential for leaching) were larger in lawns than forests (Table 3), rapid turnover of this small pool suggests that N is tightly held by the system. The average turnover time for the NO_3^- pool in lawns was <4 hours and not significantly different from the $\sim$5.5-hour turnover time in forests (Table 3). These results, paired with the high retention of ^{15}N tracer when compared to forests, suggest that fertilized lawns can be as retentive of N deposition as forested systems. A major question is how long lawns can continue to sequester high inputs of N. Previous research suggests that lawns may decrease in their capacity to retain N as they age, thereby increasing the potential for over-fertilization (see, e.g., Porter et al. 1980, Frank et al. 2006).

Accounting for nitrogen losses

Both lawns and forests saw declines in ^{15}N recovery over the course of one year (Fig. 1), but the precise mechanism for these losses was not clear. Gaseous losses

(via denitrification) are likely; however, our attempts to quantify the magnitude of denitrification losses were unsuccessful. The high temporal heterogeneity of denitrification coupled with the need for rates high enough to enrich a large background N_2 pool likely played a role in this lack of detection.

Horgan et al. (2002*a*, *b*) were able to measure direct gaseous losses of N (as N_2 and N_2O) from turfgrass using much larger amendments of $^{15}N\text{-}NO_3^-$ (49 kg N/ha compared to the 0.3 kg N/ha used in our present study). They found temporally variable fluxes of N_2 and N_2O, with an exceptionally large pulse of denitrification after a major rainfall event, suggesting the possibility for large gaseous losses. It is also possible that a significant portion of the labeled N was incorporated into soil pools below 10-cm-depth, however the results of other forest and lawn tracer studies suggest that recovery in soil and roots below 10 cm may be small (Horgan et al. 2002*a*, Engelsjord et al. 2004, Nadelhoffer et al. 2004). Soil leaching is another potential avenue for N losses, though numerous studies of N leaching in turfgrass systems suggest that these losses are usually small, but they can be significant in some cases (Petrovic 1990, Engelsjord et al. 2004). Other Baltimore (Maryland, USA) LTER research is addressing this flux.

The results of this study suggest potentially high retention of atmospheric-N deposition in lawns, which are a dominant cover type in urban areas. They lend insight into recent watershed-level research that reveals unexplained and unexpectedly high N retention in urban and suburban catchments (e.g., Groffman et al. 2004, Wollheim et al. 2005). The principal mechanisms responsible for retention of atmospheric-N deposition in lawns appear to change over time, from initial (largely transient) retention in mineral SOM, to short-term biotic uptake by plants and microbes, to eventual incorporation into mineral SOM pools. However, much remains unknown about N cycling in urban and suburban watersheds, including the long-term (decadal-scale) fate of N that is currently being retained and the capacity for continued N retention in the future. The influence on N retention of factors such as lawn age, lawn management practices, soil-disturbance history, soil compaction, soil type, and seasonal timing of deposition also require further exploration.

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