Water Air Soil Pollut (2007) 185:323–333
DOI 10.1007/s11270-007-9456-3

Potential Nitrate Leaching Under Common Landscaping Plants

José A. Amador · Richard J. Hull · Erika L. Patenaude · John T. Bushoven · Josef H. Görres

Received: 8 December 2006 / Accepted: 15 June 2007 / Published online: 13 July 2007

Abstract Studies on N losses from ornamental plantings – other than turf – are scant despite the ubiquity of these landscaping elements. We compared pore water NO_3 and extractable soil NO_3 and NH_4 in areas with turf, areas with seven different types of ornamental landscape plantings, and a native woodland. Turf areas received annual N inputs of ~48 kg ha^{-1} and annual flowers received ~24 kg N ha^{-1} at the time of planting. None of the other areas were fertilized during the course of the study. Data were collected on 23 occasions between June 2002 and November 2003. Pore water NO_3 concentrations at a 60-cm depth – based on pooled data – were highest (1.4 to 7.8 mg NO_3–N l^{-1}) under ground covers, unplanted-mulched areas, turf, deciduous trees, and evergreen trees, with no differences among these vegetation types. Lower values were observed under woodlands, annual and perennial flowers, and evergreen and deciduous shrubs. Pore water NO_3 concentrations exceeded the drinking water regulatory limit of 10 mg NO_3–N l^{-1} under ground covers, turf and unplanted-mulched areas in 39, 20 and 10% of samples, respectively. Leaching losses of NO_3–N over 18 months ranged from 0.17 kg N ha^{-1} in the woodlands to 34.97 kg N ha^{-1} under ground covers. Annual NO_3 losses under unplanted-mulched areas and ground covers were approximately twice the average N input (10 kg N ha^{-1} year^{-1}) from atmospheric deposition. Extractable NO_3 in woodland soils (0.5 μg NO_3–N g^{-1}) was lower than for all other vegetation types (3.1–7.8 μg NO_3–N g^{-1}). Extractable NH_4 levels were highest in woodlands, deciduous trees, and annual flowers (6.7–10.1 μg NH_4–N g^{-1}). Most vegetation types appear to act as net N sinks relative to atmospheric inputs, whereas unplanted-mulched areas and areas planted with ground covers act as net sources of NO_3 to groundwater.

Keywords Extractable soil N · Landscaping vegetation · Nitrate leaching · Turf

Abbreviations
AF annual flowers
PF perennial flowers
DT deciduous trees
DS deciduous shrubs
ET evergreen trees

J. A. Amador (✉) · E. L. Patenaude · J. H. Görres
Laboratory of Soil Ecology and Microbiology,
024 Coastal Institute in Kingston,
University of Rhode Island,
Kingston, RI 02881, USA
e-mail: jamador@uri.edu

R. J. Hull · J. T. Bushoven
Department of Plant Science, University of Rhode Island,
Kingston, RI 02881, USA

J. T. Bushoven
Currently at Department of Plant Science,
California State University,
Fresno, CA 93740, USA

ES evergreen shrubs
GC ground covers
T Turf
W native woodland
UM unplanted-mulched areas
OM organic matter

1 Introduction

Aesthetically pleasing, managed landscapes contribute to the quality of life in urban, suburban, and rural settings. These landscapes may include large open spaces like parks, ornamental gardens, golf courses or other communal sports facilities, as well as smaller planting assemblages, such as typical home gardens. Ornamental plantings may cover relatively large areas, contiguously or as the multi-yard mosaics typical of residential subdivisions. Despite their prevalence, little is known about NO_3 leaching from such managed landscapes.

Because of the predominance of lawn coverage in most ornamental landscapes, water quality research has focused on NO_3 leaching from the turf-soil ecosystem. In a review on the fate of nitrogen applied to turf, Petrovic (1990) concluded that only a small amount of fertilizer nitrogen (<10%) normally leaches from established turf to groundwater, a finding generally mirrored by other studies (Gold et al. 1990; Guillard and Kopp 2004). Jiang et al. (2000) have shown that even when the grass is killed, turf sites retain 90% of their accumulated nitrogen during the ensuing year even if no vegetation is replanted. When turf is re-established soon after death, a normal nitrogen retention pattern is restored within three months of reseeding (Bushoven et al. 2000).

To assess fully the environmental impact of residential, institutional or municipal landscaping, all components of the landscape must be evaluated for their contribution to leaching losses of nitrate. In a plot study of nitrogen mobility in landscape plantings in southern Florida Erickson et al. (2001) reported that plantings of mixed species leached ten times more NO_3 than did areas planted to St. Augustine grass [*Stenotaphrum secundatum* (Walt.) Kuntze] turf (48.3 vs 4.1 kg NO_3–N ha^{-1}). The study was conducted on recently established plots (<1 year old) that were fertilized at 300 and 150 kg N $ha^{-1}year^{-1}$ for the grass and mixed-species, respectively. While instructive, this study may not reflect the NO_3 mobility from landscapes maintained under minimum fertility in more northern latitudes that differ in climate, soils, and plant materials.

Little is known about the dynamics of soil carbon and nitrogen pools in landscape plantings, both factors that likely contribute to the cycling and retention of nitrogen. In their studies of carbon sequestration in the turf-soil ecosystem of Colorado golf courses, based on an analysis of historic soil-testing data, Qian and Follett (2002) found that organic matter accumulated rapidly for 25 to 30 years at rates approaching 907 kg ha^{-1} $year^{-1}$. Soil organic matter content reached 4 to 5% of dry soil mass within that period. This would be equivalent to approximately 3,000 kg organic N ha^{-1}. In a follow-up study using a grassland ecosystem model, Qian et al. (2003) determined that if clippings were retained on turf for 30 years and fertilizer rates were not reduced, significant NO_3 leaching would occur. Similar results were reported by Porter et al. (1980) in a study of the N content of variously aged lawns in New York. Currently there are no published studies on the nitrogen mobility of landscape plant communities other than turf. For turf, the evidence to date suggests that nitrogen retention may be linked to the maturity of the turf-soil ecosystem, as has been observed in forest ecosystems (Emmett et al. 1994).

The main objective of our research was to compare the concentration and leaching losses of NO_3 under different types of ornamental vegetation. The study was conducted using fully established plantings within a landscape managed according to a minimum maintenance schedule in the northeastern USA.

2 Study Area

The study was conducted in the Horticultural Display Garden and the Learning Landscape, both on the Kingston campus of the University of Rhode Island. These areas were established between 1994 and 1995 on land that had been a landscaped site for at least 50 years. A native wooded area approximately 0.5 km from the Learning Landscape was also included for comparison. The soil in the landscaped and wooded areas is mapped as a Scio very stony silt loam (coarse-silty, mixed, active, mesic Aquic Dystrudept) (Soil Survey Staff 1981) with a 0–3% slope. The organic matter content of the soil ranged from 99 g kg^{-1} in turf to 242 g kg^{-1} in the native woodland (Table 1). Woodland soil had the lowest pH (5.2), whereas soil under other vegetation types had pH values ranging from 5.9 to 6.4 (Table 1). Soil in areas covered with turfgrass was amended with dolomitic limestone (1,952 kg ha^{-1}) once prior to seeding with Chewings fescue (*Festuca rubra* spp. *commutata L. Gaudin*) in 1994. Turfgrass areas were fertilized with sulfur-coated urea at a rate of ~48 kg N ha^{-1} once a year in the spring. Areas planted to turf were irrigated only when they showed signs of water stress (e.g. wilting and change in coloration), and the amount of water applied (<2.5 cm) was not sufficient to cause percolation, as indicated by failure to obtain water samples from suction lysimeters 1, 2, and 3 days following irrigation. Annual flower beds were fertilized with composted animal manure at a rate of ~24 kg N ha^{-1} and watered (2–3 cm) once at planting time, and were not irrigated subsequently. None of the other landscaping elements were fertilized or irrigated during the course of the study. Shredded pine bark mulch (5–7.5 cm) was applied annually to all areas not covered with turf.

Southern Rhode Island has a temperate, humid climate characterized by cold winters and warm summers (Wright and Sautter 1988; Soil Survey Staff 1981). On average the growing season extends from mid-April to early October. The average annual temperature is 10°C, with average daily temperature of −1° and 21°C in January and July, respectively. Annual snowfall averages 83 cm, with most of it occurring from December to March. The average annual rainfall in Kingston, RI is 122 cm, with a relatively uniform distribution throughout the year.

Table 1 Mean[a] values of organic matter content and pH of soil under different vegetation types

Vegetation type (abbreviation)	O.M. content (g kg^{-1}±std. dev.)	pH
Native woodland (W)	242±49	5.24
Perennial flowers (PF)	123±6	6.09
Annual flowers (AF)	139±20	6.38
Deciduous shrubs (DS)	129±26	6.02
Evergreen shrubs (ES)	166±75	6.00
Evergreen trees (ET)	169±61	6.02
Deciduous trees (DT)	165±63	5.86
Turf (T)	99±16	6.07
Unplanted-mulched (UM)	107±25	6.13
Ground covers (GC)	105±24	6.05

[a] Number of replicates as indicated in Table 2.

Areas for soil and soil water sampling were chosen to represent a range of landscape vegetation types. These included annual flowers (AF), perennial flowers (PF), deciduous trees (DT), deciduous shrubs (DS), evergreen trees (ET), evergreen shrubs (ES), ground covers (GC), turf (T), native woodland (W), and unplanted-mulched areas (UM). The number of replicates was limited by the availability of each type of landscaping element. Each replicate sampling area was in a distinctly different location in the landscape. The scientific and common names of plants included in each vegetation type, number of replicates, mean replicate area, and the species found in each area are shown in Table 2.

3 Methods

3.1 Sampling

Ceramic suction cup lysimeters (2.25–cm o. d., 7-cm long) were used to sample pore water. They were installed vertically at a depth of 60 cm and at approximately half the distance from the center to the edge of a planting (for AF, PF, DS, ES, and GC) or from the trunk to the edge of the drip line of a tree (for ET, DT, and W). Lysimeters were placed near the center of the areas planted to turf (T) or unplanted areas covered by mulch (UM). One lysimeter was installed per replicate area. Suction (−80 kPa) was applied for 1 h using a hand vacuum pump (SoilMoisture Equipment Corp., Santa Barbara, California) 1 day after rainfall events exceeding 25 mm. The pore water samples were passed through a Whatman No. 42 filter, and the filtrate analyzed for nitrate colorimetrically as described below.

Soil samples were collected monthly–when the ground was not frozen – from the top 10 cm with a 2.54-cm dia. steel core sampler within a 30 cm radius of a lysimeter. Mulch and/or leaf litter were removed from the surface before sampling. One soil sample was taken per replicate area. Soil samples were stored in sealable plastic bags at 4°C.

3.2 Analyses

Soil NO_3 and NH_4 were extracted according to the method of Keeney and Nelson (1982). The NO_3 and NH_4 concentration of soil extracts and the NO_3

Table 2 Vegetation types, mean area, and scientific and common names of plants in each replicate

Vegetation type	Mean replicate area (m^2)	Replicate no.	Scientific name of plants (common name)
Annual flowers	2	1	*Dahlia* spp. (annual dwarf dahlia), *Impatiens valleriana* (dwarf impatiens)
		2	*Zinnia elegans* (Zinnia), *Portulaca grandiflora* (rose moss)
		3	*Lobelia erinus* (blue annual lobelia)
Deciduous shrubs	11	1	*Corylopsis spicata* (spike winter-hazel)
		2	*Clethra alnifolia* (summersweet)
		3	*Syringa reticulata* (Japanese tree-lilac), *Syringa meyeri* (garden lilac)
		4	*Fothergilla gardena* (dwarf fothergilla)
Deciduous trees	89	1	*Betula papyrifera* (paper birch)
		2	*Syringa reticulata* (Japanese tree-lilac)
		3	*Metasequoia glyptostroboides* (dawn redwood)
Evergreen shrubs	16	1	*Taxus baccata* (English yew)
		2	*Ilex glabra* (inkberry)
		3	*Rhododendron catawbiense* (rhododendron)
		4	*Microbiota decussata* (Siberian carpet grass)
		5	*Kalmia latifolia* (mountain laurel)
		6	*Rhododendron 'chionoides'* (white rhododendron)
Evergreen trees	86	1	*Pinus strobus* (eastern white pine)
		2	*Thuja occidentalis* (American arborvitae)
		3	*Sciadopitys verticillata* (Japanese umbrella pine)
		4	*Pinus mugo* (dwarf pine), Sweet box (*Sarcococca hookeriana*)
		5	*Picea glauca* var. *Albertiana* (Alberta spruce)
Ground covers	5	1	*Vinca minor* (myrtle)
		2, 3	*Arctostaphylos uva-uris* (bearberry)
		4	*Ajuga reptans* (carpet bugle)
		5	*Pachysandra terminalis* (pachysandra)
Perennial flowers	6	1	*Veronica alpina* (spiked speedwell)
		2	*Cimcifuga racemosa* (black cohesh)
		3	*Hemerocallis* spp.(daylily)
		4	*Hosta*-X "Krossa regal" (hosta)
		5	*Coreopsis verticillata* (threadleaf coreopsis)
Turf	–[a]	1, 2, 3, 4	*Festuca rubra* ssp. *Commutata* (Chewings fescue), *Trifolium repens* (white clover)
Native woodland	–	1, 2, 3, 4, 5	*Dennstaedtia punctiloba* (hay-scented fern), *Similax* spp. (green brier), *Fagus grandifolia* (American beech), *Quercus velutina* (black oak), *Betula populifolia* (gray birch), *Osmunda cinnamomea* (cinnamon fern), *Lycopodium* spp. (club-moss)
Unplanted-mulched	2	–	–

[a] Not applicable.

concentration of pore water samples were determined colorimetrically using an automated nutrient analyzer (Alpkem Flow Solution IV, OI Analytical, College Station, Texas).

Soil moisture was determined gravimetrically by drying at 105°C for 24 h. Soil pH was determined using a 1:10 (vol/wt) ratio of soil to water. The mixture was allowed to equilibrate for 1 h and the pH of the solution determined using a pH meter (Denver Instrument Co., Denver, Colorado) (Hendershot et al. 1993). The organic matter content of soil samples was determined for the top 5 cm of soil by mass loss-on-ignition at 550°C for 4 h (Karam 1993). Particle size analysis of soil samples was conducted using the hydrometer method (Day 1969).

3.3 Estimation of NO_3 Leaching

Nitrate leaching was estimated for each vegetation type by multiplying the volume of water percolating through the soil by the median NO_3 concentration in pore water after the most recent leaching event (Jiang et al. 2000). To estimate flow volumes through the root zone, we used a simple capacity-based model of soil water fluxes based on Class-1 evaporation pan and rainfall data from the Kingston, Rhode Island weather station that has been in operation for 80 years. Soil was characterized as a silt loam containing 35% sand, 60% silt, and 5% clay. Estimates of volumetric moisture content at the permanent wilting point (Θ_{WP}) and field capacity (Θ_{FC}), as well as porosity and bulk density (ρ_B) were obtained as described by Saxton et al. (1986). For soil in the woodland area, these values were: Θ_{WP}=0.10; Θ_{FC}=0.32; porosity=0.69; ρ_B=0.87 g cm^{-3}. In the landscaped areas, these values were: Θ_{WP}=0.07; Θ_{FC}= 0.25; porosity=0.52; ρ_B=1.26 g cm^{-3}.

Current soil moisture, SM_t, was calculated using the equation:

$$SM_t = SM_y + PPT_y - ET_y \quad (1)$$

where ET_y is the depth of the previous day's evapotranspiration (in mm), SM_t and SM_y are moisture contents (in mm of root zone) for the current and previous simulation day, and PPT_y is the depth of precipitation that fell on the previous day (in mm). ET_y was computed using the measured Class-1 pan evaporation (in mm), E_p, K_p, the pan coefficient (with a value of 0.8, estimated using the equation of Snyder (1992)), and the crop coefficient, K_c. Values of K_c were 1.05 for turf, 0.90 for deciduous forest, 0.50, for ground covers and annual and perennial plantings and 0.75 for woody ornamentals (Aaronson et al. 1987; City of Riverside Planning Department 1994). We assumed that under drought stress, K_c would become smaller (because plants were not transpiring efficiently) by a factor σ:

$$ET_y = \sigma K_c K_p E_p \quad (2)$$

The drought adjustment factor, σ, was equal to 1 when available water was greater than 30% of the potential available water ($\Theta_{FC} - \Theta_{WP}$). It was less than 0.5 when available water was less than 30% of the potential available water. The value of σ was adjusted so that the dates on which water was present in the lysimeters coincided with model predictions of percolation below the root zone. Once a value of σ was chosen for available water contents less than 30% of potential available water, it was kept constant for the entire simulation.

Eq. 1 is only correct for moistures less than Θ_{FC}. If moisture was greater than Θ_{FC} we computed the amount of percolation, PERC (in mm), from the root zone as

$$PERC = SM_t - \Theta_{FC} \quad (3)$$

PERC was subsequently subtracted from SM_t. If SM_t was greater than saturation, we assumed that some overland flow had occurred. Thus, we subtracted half the differences between SM_t and saturation (soil porosity) from SM_t prior to calculating PERC using Eq. 3. We did not subtract the entire difference because of initial storage of precipitation water and the flat to gentle slopes at the study sites, assuming that some of the difference infiltrated later.

We did not compute the contribution of snowmelt to deep percolation, assuming that soils were frozen when snow pack was present. To estimate when soils had thawed and thus become permeable to precipitation, we assumed that complete thawing occurred when the air temperature was greater than 2°C for 10 days. In some cases, we adjusted the temperature to as high as 4°C to properly predict the onset of percolation after a period of frost.

3.4 Statistical Analyses

Because values of soil NO_3 and NH_4 and pore water NO_3 concentration were not normally distributed, median values were used to represent the central tendency of the data. Differences among vegetation types were evaluated using a one-way analysis of variance on ranks. Dunn's Multiple Range Method was used to identify differences among vegetation types ($P<$ 0.05). The Pearson Product Moment correlation was employed to examine relationships among soil NO_3 and NH_4, pore water NO_3 and soil organic matter content. Statistical analyses were conducted using version 2.03 of SigmaStat (SPSS Inc., Chicago, Illinois).

4 Results and Discussion

4.1 Pore Water Nitrate

The range of NO_3 concentration in pore water samples differed with sampling date, with no clear temporal pattern (Fig. 1). Pore water NO_3 levels exceeded the maximum contaminant level (MCL) of

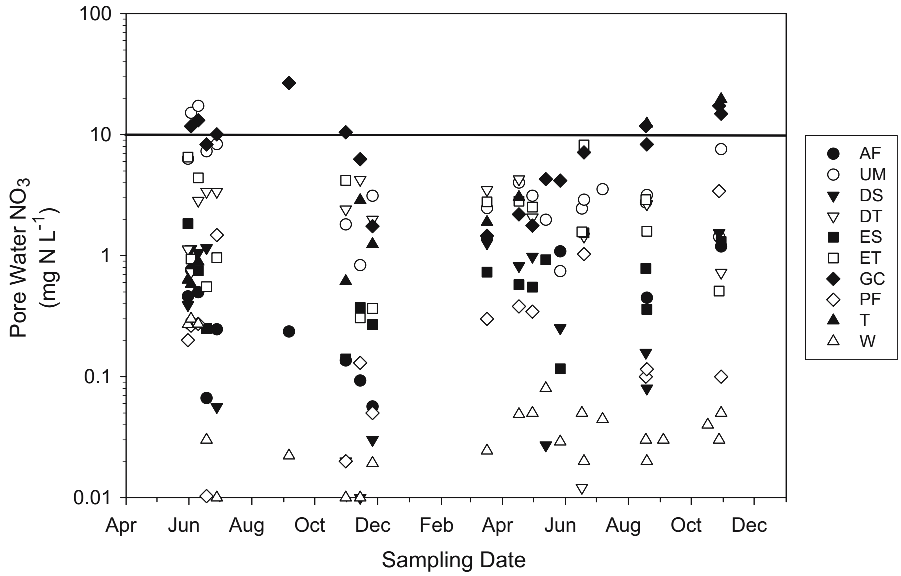

Fig. 1 Pore water NO_3 concentration under different landscaping vegetation types as a function of sampling date from June 2002 to November 2003. Values are medians ($n \geq 3$). *Bold horizontal line* represents regulatory limit for NO_3–N in drinking water (Hallberg 1989). *AF* Annual flowers; *UM* unplanted-mulched; *DS* deciduous shrubs; *DT* deciduous trees; *ES* evergreen shrubs; *ET* evergreen trees; *GC* ground covers; *PF* perennial flowers; *T* turf; *W* native woodland

10 mg NO_3–N l^{-1} for drinking water (Hallberg 1989) in three of the 10 vegetation types evaluated, with frequency of excedence following the order: GC (39%)>T (20%)>UM (10%).

Values of NO_3 concentration in pore water for all samples obtained during the course of the study for each vegetation type spanned over two orders of magnitude (Table 3). The highest median pore water NO_3 concentrations (1.4 to 7.8 mg NO_3–N l^{-1}) were observed for GC, UM, T, DT, and ET, and no differences were observed among these vegetation types. The middle range of median pore water NO_3 concentrations (0.2 to 0.3 mg NO_3–N l^{-1}) included PF, AF, DS, and ES, with no differences among vegetation types in this group. The woodlands had the lowest pore water level (0.01 mg NO_3–N l^{-1}). When the data were grouped into vegetation types with low (W), medium (PF, AF, DS, ES) and high (ET, GC, UM, T, DT) NO_3 concentration, these groups all differed from each other.

The values of pore water nitrate concentration under turf and forest vegetation in our study are comparable to those reported by others. For example, the median values of NO_3 concentration in pore water

Table 3 Median and 25 and 75th percentile values for concentration of NO_3 in pore water and of extractable soil NO_3 and NH_4 under different vegetation types

Type of vegetation	Pore water NO_3 conc. (mg N l^{-1})				Extractable soil NO_3 conc. (μg N g^{-1})				Extractable soil NH_4 conc. (μg N g^{-1})			
	n	Median	25%	75%	*n*	Median	25%	75%	*n*	Median	25%	75%
Native woodland	95	0.0 a	0.0	0.1	77	0.5 a	0.4	0.8	70	6.7 b	4.6	23.0
Perennial flowers	45	0.2 b	0.1	0.8	60	3.4 b	2.0	6.3	60	3.6 a	2.3	8.8
Annual flowers	30	0.3 b	0.1	1.1	42	3.1 b	2.2	3.9	42	10.1 b	4.9	20.3
Deciduous shrubs	47	0.3 b	0.1	1.4	63	4.8 b	2.8	9.3	61	4.8 ab	2.9	11.2
Evergreen shrubs	63	0.3 b	0.1	1.2	76	4.0 b	2.8	6.0	76	4.8 ab	2.7	10.8
Evergreen trees	52	1.4 c	0.4	4.1	68	4.3 b	2.7	7.4	68	3.6 a	2.4	7.5
Deciduous trees	40	1.8 c	0.4	4.1	45	5.0 b	2.5	10.1	46	6.8 b	3.6	13.4
Turf	34	3.0 c	0.6	10.8	70	7.8 b	3.6	16.8	65	4.7 ab	2.9	14.0
Unplanted-mulched	73	3.7 c	0.8	8.6	67	4.2 b	2.6	7.5	64	5.4 ab	2.9	12.7
Ground covers	63	7.8 c	1.7	15.4	70	5.2 b	3.2	8.8	72	4.0 a	2.4	11.4

Values are based on data pooled from June 2002 to November 2003

Values followed by the same letter within a column are not significantly different ($P<0.05$).

in the present study are within the range of values reported by Gold et al. (1990) under fertilized turf on a Merrimack sandy loam and in forested areas in Rhode Island, USA; by Jiang et al. (2000) under four different fertilized cool season turf grasses on an Enfield silt loam in Rhode Island, and by Guillard and Kopp (2004) for cool season lawn turf (*Poa pratensis*, *Lolium perenne*, and *Festuca rubra*) on a Paxton fine sandy loam in Connecticut, USA. In addition, the frequency of excedence of the MCL for NO_3–N (10 mg l^{-1}) for turf in our study (20%) is comparable to the 15% frequency observed by Guillard and Kopp (2004) for lawn turf fertilized with ammonium nitrate. The presence of N_2-fixing clover (*Trifolium repens*) in the areas under turf (~25% of the area) may explain some of the elevated pore water NO_3 concentrations, especially in late summer, when turf roots are inhibited by high temperatures (Jiang and Huang 2002) but clover roots may remain healthy.

4.2 Mass of NO_3–N Leached

Values of mass loss of NO_3–N per unit area for different vegetation types for the 18-month study period spanned two orders of magnitude, from 0.17 kg N ha^{-1} for native woodland to 34.97 kg N ha^{-1} for ground covers (Fig. 2). Vegetation types could be separated into three groups based on mass NO_3 losses: (1) low (<2 kg N ha^{-1}), which included W, ES, PF, DS, and AF; (2) medium (2 to 10 kg N ha^{-1}), which included DT, ET, and T, and (3) high (>10 kg N ha^{-1}), which included GC and UM.

When considered over the course of 1 year (June 2002 to May 2003), the mass leaching losses of NO_3–N from turf (2.81 kg N ha^{-1} $year^{-1}$) in our study are within the range observed by Guillard and Kopp (2004) for cool season lawn turf. Specifically, Guillard and Kopp (2004) reported mean losses of NO_3 ranging from 1.1 kg N ha^{-1} $year^{-1}$ for an unfertilized control treatment to 25.8 kg N ha^{-1} $year^{-1}$ for lawn turf receiving 147 kg N ha^{-1} $year^{-1}$ as ammonium nitrate. Our values are also within the range (1.9 to 9.3 kg N ha^{-1} $year^{-1}$) reported by Gold et al. (1990) for fertilized lawn turf. In addition, the annual leaching loss of NO_3–N observed in our study corresponds to 5.9% of the fertilizer N applied, within the range reported by Petrovic (1990), Gold et al. (1990) and Guillard and Kopp (2004).

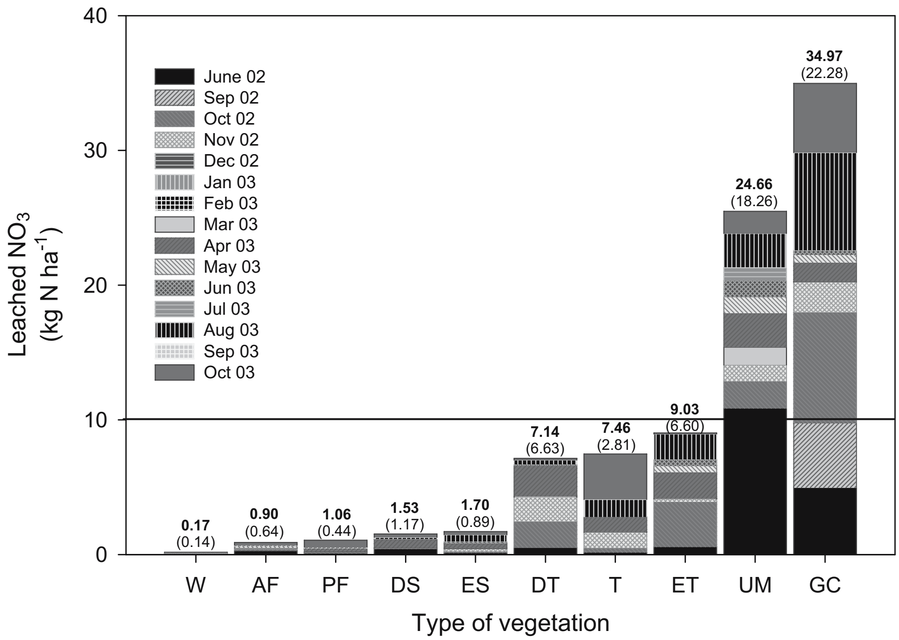

Fig. 2 Mass of NO_3 leached over the course of the study under different landscaping vegetations types. *Values in bold* represent the total mass of NO_3 leached over the 18-month study period; *values in parentheses* represent the total mass of NO_3 leached over the course of a year, from June 2002 to May 2003. *Bold horizontal line* represents annual area-weighted atmospheric N deposition estimate for the northeastern USA. (Boyer et al. 2002). *AF* Annual flowers; *UM* unplanted-mulched; *DS* deciduous shrubs; *DT* deciduous trees; *ES* evergreen shrubs; *ET* evergreen trees; *GC* ground covers; *PF* perennial flowers; *T* turf; *W* native woodland

The annual mass leaching loss of NO_3 from native woodlands in our study (0.14 kg N ha^{-1} year^{-1}) compares well with those reported by others for similar ecosystems. For example, Magill et al. (1997) reported leaching losses of NO_3 of 0.17 kg N ha^{-1} year^{-1} for a mixed hardwood forest stand at the Harvard Forest in Massachusetts, USA. Gold et al. (1990) reported a range of 1.2–1.5 kg N ha^{-1} year^{-1} for NO_3 leaching loses for a woodland with similar plant community structure on an Enfield silt loam.

The absence of data for NO_3 leaching under landscaped areas – other than turf – in the temperate, northeastern USA prevents comparison of our results to other studies. The study by Erickson et al. (2001) was conducted in southern Florida, an area with a subtropical climate and different soils and landscaping vegetation. The annual mass loss of 48.3 kg N ha^{-1} from a fertilized (50 kg N^{-1} ha^{-1}) mixed species landscape reported by Erickson et al. (2001) is nearly twice that observed in the present study for ground covers (22.28 kg N ha^{-1} year^{-1}), the vegetation type with the highest mass leaching losses of NO_3–N. Our study differs in a number of important factors from that by Erickson et al. (2001). Specifically, in the south Florida study: (1) the landscape received applications of fertilizer N, (2) the plants were grown in a mined sand medium, and (3) the plantings had been in place for less than a year. These factors likely contributed to the NO_3 leaching losses reported by Erickson et al. (2001) being considerably higher than those reported in the present study.

Our results may be examined in the broader context of nitrogen cycling at landscape scales by comparing annual mass NO_3 losses under different types of vegetation to N inputs. Aside from fertilizer inputs to turf (~48 kg N ha^{-1} year^{-1}), the landscape we studied was not fertilized, such that external N inputs were limited to atmospheric N deposition. The area-weighted average net atmospheric N deposition in the northeastern USA has been estimated at 9.59 kg N ha^{-1} year^{-1} (Boyer et al. 2002). Thus, vegetation types with values lower than inputs from atmospheric deposition (W, AF, PF, DS, ES, DT, T, and ET) can be considered net N sinks within this landscape, whereas UM and GC appear to act as net sources of NO_3 to groundwater.

4.3 Extractable Soil Nitrate and Ammonium

Extractable soil NO_3 levels followed opposite temporal trends in 2002 and 2003 (Fig. 3). For most vegetation types nitrate concentrations declined by an order of magnitude from June to December of 2002, whereas an increase of almost two orders of magnitude was observed over the same period in 2003. Soil NO_3 levels were in the range of 1 to 10 µg NO_3–N g^{-1} on most sampling dates. The woodlands were the exception, with NO_3 values consistently lower than 1 µg NO_3–N g^{-1} over the course of the study.

Median extractable soil NO_3 levels – calculated for the entire sampling period – in the woodland soil were 0.5 µg NO_3–N g^{-1}, about an order of magnitude

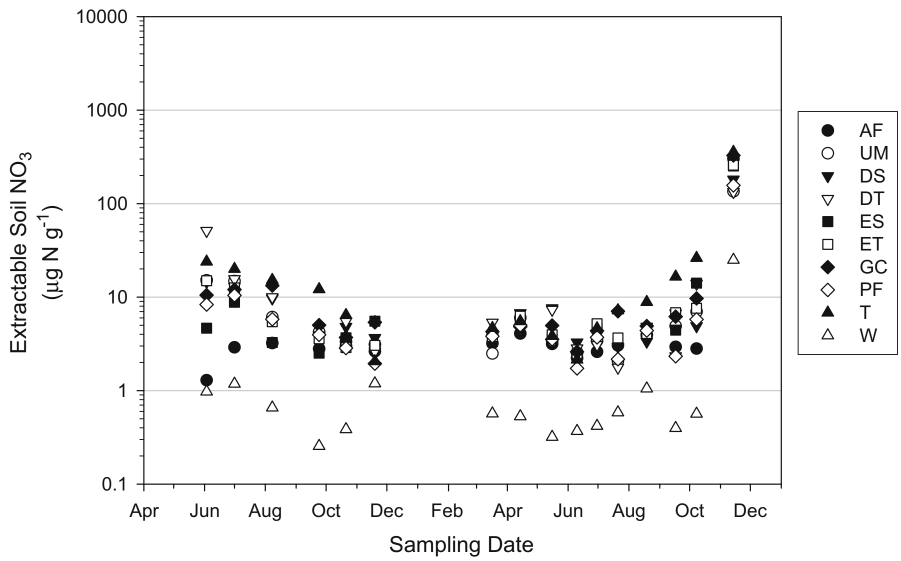

Fig. 3 Concentration of extractable NO_3 in soil under different landscaping vegetation types as a function of sampling date from June 2002 to November 2003. Values are medians ($n \geq 3$). *AF* Annual flowers; *UM* unplanted-mulched; *DS* deciduous shrubs; *DT* deciduous trees; *ES* evergreen shrubs; *ET* evergreen trees; *GC* ground covers; *PF* perennial flowers; *T* turf; *W* native woodland

lower than values for all other vegetation types (Table 3). No differences were observed among all other vegetation types, with soil NO_3 concentrations ranging from 3.1 μg NO_3–N g^{-1} for annual flowers to 7.8 μg NO_3–N g^{-1} for turf.

Values of extractable soil NH_4 generally declined about an order of magnitude from June to December 2002, and increased over two orders of magnitude from October to December 2003 (Fig. 4). As with extractable soil NO_3, NH_4 concentrations were within the range of 1 to 10 μg NH_4–N g^{-1} for most of the sampling period.

Analyses of pooled data indicated that median levels of extractable soil NH_4 ranged from 3.6 μg NH_4–N g^{-1} for perennial flowers and evergreen trees to 10.1 μg NH_4–N g^{-1} for annual flowers (Table 3). Differences were observed between those vegetation types with low NH_4 levels (PF, ET, GC) and those with high levels (W, DT, AF). The range of values observed in our study for extractable soil NO_3 (1 to 15 μg N g^{-1}) and NH_4 (3 to 11 μg N g^{-1}) in turf are comparable to those reported by Jiang et al. (2000).

Because the bulk of net NO_3 production – independent of mechanism – takes place in the upper portions of the soil profile, it is reasonable to expect a positive relationship with concentration of NO_3 in pore water. Furthermore, because NH_4 is a precursor of NO_3 production, its concentration may also be coupled to pore water NO_3 levels. A relationship between soil organic matter and extractable NO_3 and/or NH_4 may also be expected because the microbial mineralization of N associated with organic matter produces inorganic N. Correlation analyses using the Pearson Product Moment method indicated that there was a positive correlation between soil NO_3 and $\log_{10}$ pore water NO_3 (r=0.779, P=0.0079). However, no correlation was observed between either soil NO_3 or NH_4 and untransformed pore water NO_3 values. Soil NO_3 and organic matter were positively correlated (r=0.746, P=0.0101), but there was no correlation between soil NH_4 and organic matter. The exponential increase in pore water NO_3 with linear increases in soil extractable NO_3 suggest that the processes controlling NO_3 leaching from surface soil differ among vegetation types.

Some of the differential effects of vegetation type may be due to differences in root architecture. Mature trees generally have relatively shallow nutrient-absorbing root systems – aside from the deep roots involved in anchoring – that may result in less opportunity for interception of nitrate deeper into the soil profile than in vegetation with a greater concentration of fine roots at greater depths. For example, more than 90% of the small, nutrient-absorbing roots were in the top 12.5 cm of soil in a pine forest on clay soils of North Carolina (Coile 1937). Furthermore, the roots most actively involved in nutrient uptake are often found outside the drip line, whereas we took soil and water samples within the area under the tree canopy, in part to avoid encroaching on other vegetation types. Additional data on the relationship between root architecture and N removal by land-

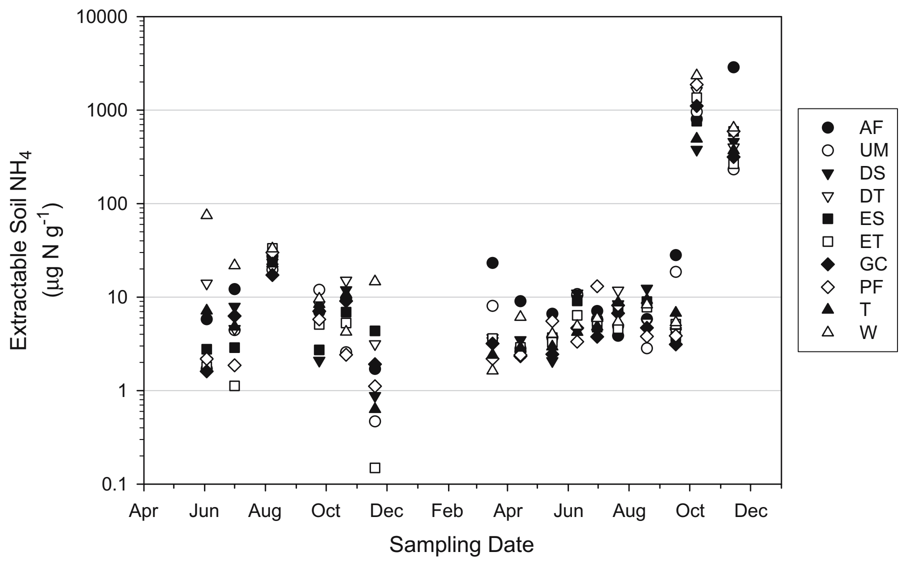

Fig. 4 Concentration of extractable NH_4 in soil under different landscaping vegetation types as a function of sampling date from June 2002 to November 2003. Values are medians (n≥3). *AF* Annual flowers; *UM* unplanted-mulched; *DS* deciduous shrubs; *DT* deciduous trees; *ES* evergreen shrubs; *ET* evergreen trees; *GC* ground covers; *PF* perennial flowers; *T* turf; *W* native woodland

scaping plants is necessary to examine this hypothesis. Nitrogen taken up by trees in a forest ecosystem may be returned to the soil via the litter, where the added N can be nitrified and leached (Pregitzer et al. 2004). However, leaf fall may not be an important factor in explaining differences in N loss among vegetation types. Litter in hardwood and conifer stands contributes only 19 and 26 kg N ha^{-1} $year^{-1}$, respectively (Kozlowski et al. 1991). This is a modest amount of N and its mineralization takes place over a 5–18 year period. Thus, leaf fall in autumn seems unlikely to contribute significantly to N leaching.

The differences observed among vegetation types in our study may also be accounted for by differences in the dynamics and relative magnitude of processes involved in N production and consumption, and the factors that control them. For example, the size and composition of the rhizosphere microbial community differs among plant species (Kennedy 2005). Such differences may be responsible for the differential effects that plant species exert on rates of nitrification and denitrification (e.g. Priha et al. 1999). Similarly, the amount and type of carbon compounds exuded by plant roots, which may fuel NO_3 removal by denitrification, differ among plant species (Kennedy 2005).

The elevated NO_3 concentrations in pore water and high amounts of leached NO_3 observed under unplanted-mulched areas are likely due to the absence of a plant sink for N produced by microbial mineralization of soil organic matter, a phenomenon observed by others (e.g. Jiang et al. 2000). The results for ground covers – for which the amount of NO_3 leached was higher than for the UM treatment – are more difficult to explain. The ground cover plants included in our study are not known to host N_2-fixing bacteria, eliminating this as an additional source of N. Alternatively, the presence of ground covers in soil may accelerate N mineralization, exceeding the capacity of plants to take up the resulting ammonium. The excess NH_4 may then be oxidized to NO_3 and be subject to leaching.

5 Conclusions

The concordance between our results and published studies – conducted under similar conditions to ours – in terms of leaching losses of NO_3 from woodlands and turf, and N retention in turf, suggests that our results likely represent what takes place in these landscapes. In addition, the use of a reasonably mature (7–8 years) landscape with minimal management inputs that was established within a larger, suburban context strengthens the transferability of our results to other similarly established landscaped areas in the region.

Our results have implications for the design and management of sustainable landscaping to maintain groundwater quality. Native woodlands clearly constitute the most benign type of vegetation with respect to potential NO_3 leaching into groundwater. As such, their incorporation in an undisturbed state into landscape designs should be given serious consideration. Annual and perennial flowers and deciduous and evergreen shrubs also contribute minimally to NO_3 leaching, and thus should be given priority in terms of area covered by plantings. Managed turf and landscape plantings of ornamental deciduous trees and evergreen trees exhibited similar mass NO_3 losses and represent an intermediate risk of NO_3 loss. In contrast, ground cover plants, and unplanted-mulched areas appear to act as net sources of NO_3 to ground water, and thus should be used sparingly.

Aesthetically pleasing landscaping and protection of groundwater quality may be achieved by minimizing disturbance of existing natural woodlands, reducing the unplanted areas and areas covered by vegetation types that are associated with high levels of pore water NO_3, and making more extensive use of those landscaping plants that show minimal pore water NO_3 levels.

Acknowledgements We thank Cheryl Tefft, Nicole Schwarz, Tracey Daly, Andrea Bixby, Maricruz Merino, Jack Barry, and Charles Gilbert for their help with different aspects of this study. This research was funded in part by a grant from the Rhode Island Water Resources Center and by the Rhode Island Agricultural Experiment Station (Contribution No. 5101).

References

Aaronson, L. J., Gold, A. J., Hull, R. J., & Cisar, J. L. (1987). Evapotranspiration of cool season turfgrasses in the humid Northeast. *Agronomy Journal, 79*, 901–905.

Boyer, E. W., Goodale, C. L., Jaworski, N. A., & Howarth, R. W. (2002). Anthropogenic nitrogen sources and relationships to riverine nitrogen export in the northeastern U. S. A. *Biogeochem, 57/58*, 137–169.

Bushoven, J. T., Jiang, Z., Ford, H. J., Sawyer, C. D., Hull, R. J., & Amador, J. A. (2000). Stabilization of soil nitrate by reseeding with perennial ryegrass following sudden turf death. *Journal of Environmental Quality, 29*, 1657–1661.

City of Riverside Planning Department. (1994). Water efficient landscaping and irrigation ordinance. Summary and Design Manual, 2nd ed.

Coile, T. S. (1937). Distribution of forest tree roots in North Carolina Piedmont soils. *Journal of Forestry, 35*, 247–257.

Day, P. R. (1969). Particle fractionation and particle-size analysis. In C. A. Black, D. D. Evans, J. L. White, L. E. Ensminger, F. E. Clark (eds.) *Methods of soil analysis. Part 1*. (pp. 545–577). Madison, WI, USA: Agronomy Monograph 9. ASA and SSSA.

Emmett, B. A., Reynolds, B., Stevens, P. A., Norris, D. A., Hughes, S., Görres, J. H., et al. (1994). Nitrate leaching from afforested Welsh catchments – Interactions between stand age and nitrogen deposition. *Ambio, 22*, 386–394.

Erickson, J. E., Cisar, J. L., Volin, J. C., & Snyder, G. H. (2001). Comparing nitrogen runoff and leaching between newly established St. Augustinegrass turf and an alternative residential landscape. *Crop Science, 41*, 1889–1895.

Gold, A. J., DeRagon, W. R., Sullivan, W. M., & Lemunyon, J. L. (1990). Nitrate–nitrogen losses to ground water from rural and suburban land uses. *Journal of Soil and Water Conservation, 45*, 305–310.

Guillard, K., & Kopp, K. L. (2004). Nitrogen fertilizer form and associated nitrate leaching from cool-season lawn turf. *Journal of Environmental Quality, 33*, 1822–1827.

Hallberg, G. R. (1989). Nitrate in ground water in the United States. In R. F. Follett (Ed.), *Nitrogen management and ground water protection*, (pp. 35–74). Amsterdam, The Netherlands: Elsevier.

Hendershot, W. H., Lalande, H., & Duquette, M. (1993). Soil reaction and exchangeable acidity. In M. R. Carter (Ed.), *Soil sampling and methods of analysis* (pp. 141–145). Boca Raton, FL, USA: Lewis Publishers.

Jiang, Y., & Huang, B. (2002). Effects of drought or heat stress alone and in combination on Kentucky bluegrass. *Crop Science, 40*, 1358–1362.

Jiang, Z., Bushoven, J. T., Ford, H. J., Sawyer, C. D., Amador, J. A., & Hull, R. J. (2000). Mobility of soil nitrogen and microbial responses following the sudden death of established turf. *Journal of Environmental Quality, 29*, 1625–1631.

Karam, A. (1993). Chemical properties of organic soils. In M. R. Carter (Ed.), *Soil sampling and methods of analysis* (pp. 459–485). Boca Raton, FL, USA: Lewis Publishers.

Keeney, D. R., & Nelson, D. W. (1982). Nitrogen-Inorganic forms. In A. L. Page, et al. (Eds.), *Methods of soil analysis. Part 2*. 2nd ed. (pp. 643–698), Madison, WI, USA: Agronomy Monograph 9. ASA and SSSA.

Kennedy, A. C. (2005). Rhizosphere. In D. M. Sylvia, J. J. Fuhrmann, P. G. Hartel, & D. A. Zuberer (eds.), *Principles and applications of soil microbiology*, 2nd ed. (pp. 242–262). Upper Saddler River, NJ, USA: Pearson-Prentice Hall.

Kozlowski, T. T., Kramer P. J., & Pallardy, S. E. (1991). *The physiological ecology of woody plants*. San Diego, CA, USA: Academic Press.

Magill, A. H., Aber, J. D., Hendricks, J. J., Bowden, R. D., Melillo, J. M., & Steudler, P. A. (1997). Biogeochemical response of forest ecosystems to simulated chronic nitrogen deposition. *Ecological Applications, 7*, 402–415.

Petrovic, A. M. (1990). The fate of nitrogenous fertilizers applied to turfgrasses. *Journal of Environmental Quality, 19*, 1–14.

Porter, K. S., Bouldin, D. R., Pacenka, S., Kossack, R. S., Shoemaker, C. A., & Pucci, A. A. Jr. (1980). *Studies to assess the fate of nitrogen applied to turf: Part I. Research Project Technical Completion Report OWRT Project A-086-NY.* Ithaca, NY, USA: Cornell University.

Pregitzer, K. S., Zak, D. R., Burton, A. J., Ashby, J. A., & MacDonald, N. C. (2004). Chronic nitrate additions dramatically increase the export of carbon and nitrogen from northern hardwood ecosystems. *Biogeochem, 68*, 179–197.

Priha, O., Hallantie, T., & Smolander, A. (1999). Comparing microbial biomass, denitrification enzyme activity, and numbers of nitrifiers in the rhizospheres of *Pinus sylvestris*, *Picea abies*, and *Betula pendula* seedlings by microscale methods. *Biology and Fertility of Soils, 30*, 14–19.

Qian, Y., & Follett, R. F. (2002). Assessing soil carbon sequestration in turfgrass systems using long-term testing data. *Agronomy Journal, 94*, 930–935.

Qian, Y., Bandaranayake, W., Parton, W. J., Mecham, B., Harivandi, M. A., & Mosier, A. R. (2003). Long-term effects of clipping and nitrogen management in turfgrass on soil organic carbon and nitrogen dynamics: The CENTURY model simulation. *Journal of Environmental Quality, 32*, 1694–1700.

Saxton, K. E., Rawls, W. J., Romberger, J. S., & Papendick, R. I. (1986). Estimating generalized soil-water characteristics from texture. *Soil Science Society of America Journal, 50*, 1031–1036.

Snyder, R. L. (1992). Equation for evaporation pan to evapotranspiration conversions. *Journal of Irrigation and Drainage Engineering, 118*, 977–980.

Soil Survey Staff. (1981). *Soil survey of Rhode Island. USDA-Soil Conservation Service*. Washington, DC, USA: US Govt. Print. Office.

Wright, W. R., & Sautter, E. R. (1988). *Soils of Rhode Island landscapes*. Rhode Island Agricultural Experiment Station Bulletin. No. 429, Kingston, RI, USA.