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Short- and long-term effects of site factors on net N-mineralization and nitrification rates along an urban-rural gradient

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Abstract. Long- and short-term effects of urban site factors on net N-mineralization and nitrification rates were investigated in oak stands along an urban-rural land-use transect in the New York City metropolitan area. We used reciprocal transplants of undisturbed soil cores between urban and rural forests to determine the relative importance of long-term effects (mor vs. mull soils, quality of soil organic matter, and deposition of N) vs. short-term effects (soil temperature) of urban factors in controlling field N-transformation rates along the gradient. In addition, undisturbed soil cores from surface (A, Oe horizons) and subsurface (B horizon) soil were collected from urban, suburban, and rural stands and allowed to incubate in these respective sites to compare the net effect of all urban factors with transplanted-core results. The transplant experiment revealed that soil type (long-term) affected net N-mineralization and nitrification rates. Urban soils nitrified nearly 6.3 and 5.4 times more than rural soils incubating in urban and rural stands, respectively ($p = 0.003$ and $p = 0.002$, respectively). Similarly, in rural stands total accumulation of inorganic N was 87% higher in urban than in rural soils, whereas in urban stands, urban soils mineralized 83% more N than rural soils ($p = 0.043$ and 0.08, respectively). Comparing soils incubating in their native locations, urban soils incubating in urban stands mineralized more than 2.5 times the amount of N than rural soils incubating in the rural stands ($p = 0.019$). By contrast, urban soils incubating in urban stands exhibited a 8-fold increase in nitrification over rural soils incubating in rural stands ($p = 0.008$). As with the transplanted cores, the urban and suburban environments had a positive effect on net rates of N-mineralization and nitrification in both surface and subsurface layers of soil. The surface layer of suburban and urban stands had a 3- and 2.3-fold higher accumulation of net inorganic N than rural stands (ANOVA, $p = 0.05$). Similarly, in the subsurface layer both urban and suburban stands had 2.6-fold higher net N-mineralization rate than rural stands (ANOVA, $p = 0.01$). Along this urban-rural gradient, soils in oak stands exhibit higher net nitrification and, to a lesser extent, net N-mineralization rates in urban and suburban stands than in rural stands. Results from the transplant experiment and *in situ* measurements of surface and subsurface soil indicate that long-term effects (mor vs. mull soils, N deposition) contribute to the higher N-transformation rates in urban and suburban stands. As a result of these effects, urban and suburban stands have the potential for higher losses of N than rural stands.

Keywords: N-mineralization, nitrification, earthworms, exotic species, forests, buried bags, soil organic matter, urban

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Introduction

Less than 2% of the total N in forest ecosystems is released as inorganic N on an annual basis (Mellilo *et al.*, 1982). It is this proportion of total N that is readily available for plant uptake. For this reason, N often is limiting in forest ecosystems and thus is the primary factor controlling the production of aboveground biomass (Vitousek and Howarth, 1991). Under conditions in which N inputs exceed biological demand, the production of nitrate, or nitrification, may increase with the eventual loss of nitrate from the system (Vitousek *et al.*, 1982; Robertson, 1982; Aber *et al.*, 1989). As a result, nitrification is an important process that predisposes ecosystems to N loss (Zhu and Carreiro, 1999).

Nitrate losses have been associated with forest site disturbances in which plant removal reduces biological demand for N, while the onset of new site conditions generally increases ammonification rates (Likens *et al.*, 1969). By contrast, long-term additions of inorganic N in previously disturbed forest plots indicate that more N can be accumulated in sites in which the disturbance resulted in N losses, e.g., cultivation (Aber *et al.*, 1998). These results suggest the importance and complex nature of both site history and disturbance on N-transformation rates in forest soils.

Modifications in N-mineralization and nitrification rates also have been recorded in polluted environments with the nature of the response depending on the type and flux of the pollutants involved (Pouyat *et al.*, 1997). In heavily metal-polluted areas such as those adjacent to smelters, N-transformation rates have been low compared to those in adjacent non-polluted areas (Tyler, 1975; Jackson and Watson, 1977). In major metropolitan regions receiving high rates of N deposition, N-transformation rates have been high (Fenn and Dunn, 1989; Fenn, 1991; Kuperman, 1999). Laboratory experiments have largely verified the results of the smelter studies in which additions of high concentrations of heavy metals and artificial irrigation of acidified water generally reduce N transformation rates (see reviews by Smith (1990) and Bääth (1989)). However, with amendments of moderate concentrations of heavy metals, N-transformation rates could increase (Bääth, 1989). In contrast to heavy metals, inorganic N additions in laboratory incubations generally have failed to affect net rates of nitrification and or ammonification (Zhu and Carreiro, 1999; Ste-Marie and Paré, 1999). For nitrification, the absence of a short-term response to N amendment may be explained by a lack of nitrifying bacteria living in the soil or by the relatively long generation times of autotrophic nitrifying bacteria (Killham, 1994).

Contrasting results for N-transformation rates have been reported for forest soils embedded within urban areas. Inman and Parker (1978) and White and McDonnell (1988) found lower decomposition and N-transformation rates in urban forest soils compared to those in rural forest soils in the same region. However, N-transformation rates in oak-dominated forest stands along an urban-rural gradient were greater in urban than in rural stands (Pouyat *et al.*, 1997; Zhu and Carreiro, 1999).

In this study, we build upon comparative research on the effects of urban, suburban, and rural environments on forest ecosystem structure and function along an urban-rural transect in the New York City metropolitan area (McDonnell and Pickett, 1990; McDonnell *et al.*, 1993; Pouyat *et al.*, 1995). Measurements along this transect suggest a soil environmental and atmospheric deposition gradient in oak stands embedded within urban, suburban, and

rural land uses. Concentrations of heavy metals and temperatures to a 10-cm depth were higher in urban stands; soil texture and bulk density did not differ statistically along the transect (Pouyat *et al.*, 1995). While heavy metal concentrations were relatively high in the urban forest soils, the levels do not exceed toxicity thresholds for most soil organisms (Pouyat *et al.*, 1997). Throughfall measurements beneath oak canopies showed up to a 3-fold greater total N deposition in the urban than in rural stands (Lovett *et al.*, 2000). In addition, nonnative species of earthworms were more numerous and higher in biomass in urban than in rural stands (Steinberg *et al.*, 1997).

The net result of the changes in soil chemistry, temperature, and earthworm abundances was that potential net N-mineralization and nitrification rates of the mineral soil were high in the urban, intermediate in the suburban, and low in the rural stands (Pouyat *et al.*, 1997). By contrast, decay rates measured in lab and field incubations along the transect demonstrated that leaf litter of red oak (*Quercus rubra* L.) collected from the rural stands decomposed and mineralized N more rapidly than suburban and urban red oak leaves (Carreiro *et al.*, 1999; Pouyat and Carreiro, 2003). These results suggest that while urban site factors such as high N deposition, earthworm abundance, and soil temperature tend to accelerate leaf litter decay and N-transformation rates, low quality urban litter and soil organic matter tend to reduce decay rates and N availability in these soils.

The purpose of this study was to investigate the long- and short-term effects of urban site factors on net N-mineralization and nitrification rates among oak stands along the land-use transect described previously. Specifically, we used reciprocal transplants of undisturbed soil cores between urban and rural forests to determine the relative importance of long-term, or site history, effects (mor vs. mull soils, quality of soil organic matter, and atmospheric N deposition) vs. short-term effects (soil temperature) of urban environmental factors in controlling field N-transformation rates along the land-use gradient. In addition, undisturbed soil cores were collected from urban, suburban, and rural stands and allowed to incubate in these respective sites to compare the net effect of all urban site factors with the transplanted core results. For these field incubations, some cores were separated into surface (A, Oe horizons) and subsurface (upper B horizon) soil prior to conducting inorganic N measurements. We conducted a separate analysis for each layer to avoid the mixing of mineral and organic soil layers and to account for variations in the depth of organic-matter concentration due to earthworm activity in urban and some suburban stands.

Methods

Site descriptions

In a previous study (Pouyat and McDonnell, 1991), a 20-km-wide by 130-km-long belt transect was established along an urban-rural land-use gradient in the study area (figure 1). Forest stands were selected using the following criteria: (1) location on upland sites on either of two soil series (Hollis and Charlton), both of which are classified as well-drained, moderate to shallow, sandy loam inceptisols in the Dystrochrepts group that vary only in depth to bedrock (Gonick *et al.*, 1970; Hill *et al.*, 1980); (2) oak-dominated forest with *Quercus rubra* L. and *Q. velutina* Lam. as major components of the overstory (combined

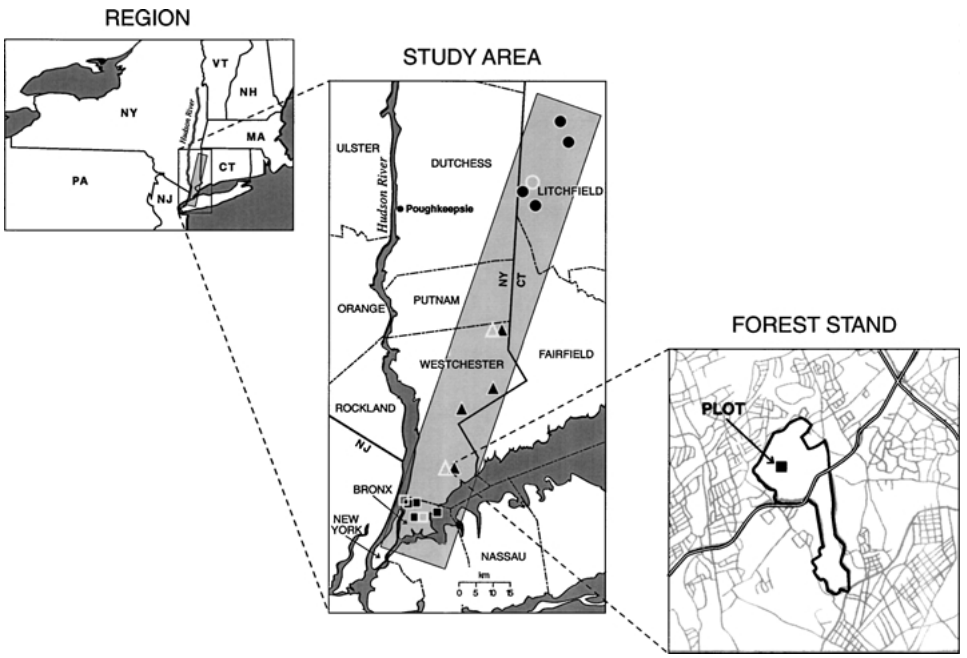

Figure 1. Location of the urban-rural transect in the New York metropolitan area. Transect runs from highly urbanized Bronx, New York, to rural Litchfield County, Connecticut. Twelve plots in oak-dominated stands (solid) were established along the transect to conduct the reciprocal transplant study. Urban, suburban, rural plots are depicted as squares, triangles, and circles, respectively. Five additional plots (open) were established along with the original 12 plots to incubate cores as surface (Oe and A horizons) and subsurface (upper B horizon) soils in 1993.

basal area of 21–47 m^2 ha^{-1}; M.L. Cadenasso, M.J. McDonnell, and S.T.A. Pickett, unpublished data); (3) minimum stand age of 70 years; and (4) no visual evidence of natural disturbance and no documented evidence of human impact, such as logging, for at least 70 years. By design, none of the study sites had a significant proportion of nonnative tree species. However, on the basis of previous measurements, there were significant differences in the abundance of nonnative species of earthworms and characteristics of the soil organic horizon between plots. On average, the urban, suburban, and rural plots had 25.0, 7.0, and 2.5 individuals m^{-2} of non-native species of earthworms, respectively (Steinberg *et al.*, 1997). These include individuals of the families Megascolecidae (*Amynthas agrestis* and *A. hawayanus*) and Lumbricidae (*Lumbricus rubellus* and *Dendrobaena octaedra*). Moreover, all of the urban plots and several suburban plots have mull soil organic horizons (Table 1), which is typical for forest soils with high earthworm densities. The study area is described in detail in Pouyat (1992), Medley *et al.* (1995), and Pouyat *et al.* (1995).

Originally, 27 oak forest plots were located along the transect. Each plot was 20 × 20-m and was divided into sixteen 5 × 5-m (0.0025 ha) quadrats. In previous studies, soil characteristics, soil organism abundances, and decomposition rates of a reference litter type were measured (Pouyat *et al.*, 1994, 1995, 1997; Steinberg *et al.*, 1997). For this study, a subset of 8 plots was selected randomly from the original 27 (see Carreiro *et al.*, 1999) to conduct a

Table 1. Description of soils in 17 sites along urban-rural land-use gradient in New York City metropolitan area. Values are the mean of 4 composited samples (10-cm depth) for each plot. Standard errors are given in parentheses. Plot R1 surface 10-cm depth sample included Oe horizon. Adapted from Pouyat (1992) and Zhu and Carreiro (1999)

Land-use type	Site	Site description	pH	SOM (%)	Organic C ($g\ kg^{-1}$)	Total N ($g\ kg^{-1}$)	Sand (%)	Clay (%)
Urban	N2	Mull/mor soil, earthworms present	4.23	10.9 (0.2)	6.8 (0.2)	0.28 (0.01)	71.5 (1.5)	9.9 (0.7)
	N3	Mull soil, mixed by earthworms	4.45	9.4 (0.2)	5.4 (0.4)	0.27 (0.02)	72.9 (0.7)	9.7 (0.9)
	P1	Mull soil, mixed by earthworms	4.40	12.1 (0.3)	9.0 (1.4)	0.36 (0.03)	71.7 (1.1)	9.6 (1.3)
	R1	Mull/mor soil, earthworms present	4.40	22.1 (0.1)	NA	NA	NA	NA
	V2	Mull soil, mixed by earthworms	4.61	6.7 (0.3)	3.2 (0.2)	0.17 (0.01)	80.6 (0.9)	7.3 (0.9)
	V3	Mull soil, mixed by earthworms	4.85	8.3 (0.5)	4.8 (0.4)	0.21 (0.02)	74.3 (1.3)	9.0 (0.5)
Suburban	ML3	Mor soil, earthworms present	4.41	6.5 (0.2)	3.9 (0.7)	0.26 (0.05)	78.1 (0.9)	8.7 (0.6)
	ML4	Mor soil, 1 cm O layer	4.52	6.9 (0.3)	3.0 (0.3)	0.19 (0.01)	NA	NA
	MR3	Mull soil, mixed by earthworms	4.76	8.8 (0.4)	3.9 (0.7)	0.26 (0.05)	79.8 (0.3)	6.3 (0.7)
	MR4	Mull/mor soil, 1 earthworms present	4.60	7.9 (0.1)	3.0 (0.2)	0.20 (0.01)	77.9 (0.9)	8.7 (0.5)
	S1	Mor soil, 1 cm O layer	4.50	9.3 (0.5)	4.6 (0.4)	0.22 (0.02)	70.5 (2.7)	12.6 (0.8)
	S2	Mull/mor soil, earthworms present	4.55	11.4 (0.7)	5.3 (0.6)	0.27 (0.02)	67.0 (1.5)	13.7 (1.0)
Rural	H1	Mor soil, 1-2 cm O layer	4.70	7.02 (0.8)	5.7 (1.0)	0.26 (0.04)	71.4 (2.1)	11.0 (0.4)
	H3	Mor soil, 1-2 cm O layer	4.52	4.9 (0.2)	2.5 (0.5)	0.11 (0.01)	74.1 (2.7)	7.5 (2.5)
	MF1	Mor soil, 1-2 cm O layer	4.52	12.1 (0.7)	6.1 (0.4)	0.27 (0.02)	72.1 (1.0)	10.6 (1.1)
	MF3	Mor/mull soil, earthworms present	4.75	7.6 (0.4)	6.1 (0.8)	0.25 (0.02)	75.3 (0.5)	9.7 (0.7)
	MSP4	Mor soil, 1-cm O layer	4.80	10.5 (0.4)	NA	NA	NA	NA

reciprocal transplant experiment of undisturbed soil cores between urban and rural land-use types (figure 2). In addition, four suburban stands were sampled, though the cores were not reciprocally transplanted with the urban and rural stands (8 transplant and 12 native plots). The soil core transplants were sampled and incubated between April and December, 1990.

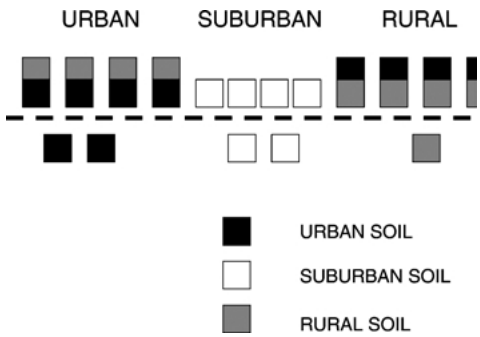

Figure 2. Schematic diagram of soil core reciprocal transplant study (1990) and surface (A and Oe horizons) and subsurface (upper B horizon) soil incubations (1993) along an urban-rural transect in the New York City metropolitan area. The 12 plots depicted above the dashed line were included in the reciprocal transplant study and the 5 plots below the dashed line were added in 1993. Each square represents a plot located in an urban, suburban, or rural forest stand.

The urban plots were located in the New York Botanical Garden Forest (N2, N3), Van Cortlandt Park (V3), and Pelham Bay Park (P1) in the Bronx, New York. The rural plots were located in Housatonic State Forest (H1, H3) and Mohawk State Forest (MF1, MF3) in Litchfield County, Connecticut. The suburban plots were located in Saxon Woods County Park (S1 and S2), Mountain Lakes County Park (ML3), and Mianus River Gorge (MR3), Westchester County, New York. Five plots were added to the transplant study plots between April and December 1993, to measure *in situ* rates of N-mineralization and nitrification for surface (Oe and A horizons) and subsurface (upper B horizon) soils along the entire urban-rural transect (17 plots total). Additional urban plots were located in Van Cortlandt (V2) and River Dale Park (R1) in the Bronx and Manhattan, New York, respectively. A rural plot was added in Macedonia State Park, Connecticut (MP4). Two suburban plots were added at Mountain Lakes (ML4) and Mianus River Gorge (MR4) Parks, Westchester County, New York.

Potential rates

Potential rates were measured concurrently with *in situ* measurements in the reciprocal transplant experiment using 8-week laboratory incubations. To obtain soil for laboratory incubations, the surface 5 cm of soil was sampled in early April 1990. Prior to taking a sample, we removed the litter layer (Oi) from the soil surface. Three 5 × 5-cm cores were removed from 4 of 16 quadrats (5 × 5-m or 0.0025 ha) in each plot, composited by quadrat, placed in Whirl-pak® bags, and stored on ice or refrigerated before analysis (always within 24 hr). The samples were sieved to remove material greater than 6.4 mm (0.25 inch) while preserving crumb and granular soil structure. The concentration of organic matter was determined on a subsample by loss upon ignition at 450°C after 4.5 hr.

Potential N-mineralization and nitrification rates were determined using incubation under aerobic conditions at standard moisture (50% of water-holding capacity) and temperature (20°C) conditions (White, 1986). Water-holding capacity was determined by placing

subsamples of sieved soil into stoppered funnels and adding water until the subsample was covered with water. After 30 minutes, the subsamples were drained by gravity for 1 hr and the water content determined by the mass lost after heating at 105°C to a constant weight. The water-holding capacity equaled the amount of retained water (White and McDonnell, 1988). Subsamples were then placed in plastic cups; each cup contained approximately 10 g dry-weight soil. Two subsamples of each sample were extracted immediately with 100 mL of 2 M potassium chloride. After settling for 24 hr, the potassium chloride extracts and method blanks were filtered to remove floating material. The remaining subsamples were covered with plastic wrap to minimize water loss but allow CO_2 and O_2 gas exchange (Bremner and Douglas, 1971) and incubated at 20°C. Two cups of each sample were removed and extracted with 100 mL potassium chloride and composited for analysis at 4 and 8 weeks. The filtrate was analyzed for ammonium by an automated phenolate method (Technicon AutoAnalyzer Industrial Method # 19-69) and nitrate by an automated nitroprusside method (Industrial Method # 33-69W).

Potential net nitrification rates were calculated as the change in extractable NO_3^--N concentrations during incubation. Potential net N-mineralization was the change in the sum of NO_3^--N and NH_4^+-N concentrations during incubation. Denitrification and ammonia volatilization were assumed to be negligible during the incubation period. Data on potential net N-mineralization and nitrification are reported on a per gram organic matter weight (g^{-1} OM) and per cubic centimeter of soil (cm^{-3}) basis. Reporting data on N-mineralization on an OM basis takes into consideration any variation that may occur among sites in the concentration of soil organic matter. Reporting data on N-mineralization on a volume basis takes into consideration any variation in soil bulk density that may occur among sites.

We used a repeated measures MANOVA (SAS, 1987) to statistically test for differences between the inorganic N accumulation curves of the urban, suburban, and rural land-use types over the 8-week incubation period. A repeated measures test was used as an alternative to treating time as a split unit factor, because samples for incubations were collected repeatedly from the same plot or unit. The treatment of time as a split unit factor makes it difficult to interpret data obtained when the sample of experimental material at each time is different (Mead, 1990). In this procedure, a MANOVA tests for changes in N-related variables over time and interactions between two types of effects (land-use type and time).

Soil core transplants

Net N-mineralization and nitrification rates were measured in the field using polyethylene bag incubations (Eno, 1960). The *in situ* incubations were initiated at the same time incubations began in the laboratory. To measure initial pools of NH_4^+-N and NO_3^--N in the transplant plots (figure 2), 8 undisturbed cores (2 from each quadrat) were collected from each stand and were returned to the original source stand to incubate for all sample locations (urban and rural stands) along the transect. We refer to these cores as native soil. To separate long- from short-term site effects on net N-mineralization and nitrification rates, reciprocal transplants of soil were made between plots at the extreme ends of the environmental gradient (urban and rural stands; figure 2). We refer to these cores as soil transplants. Four 5 × 5-cm cores, each of native and transplanted soil, were placed in each of the 4 randomly selected quadrats

per plot in early April 1990 resulting in 32 sample locations (8 plots × 4 quadrats) along the gradient. In each quadrat, 2 cores were placed in polyethylene bags, sealed, and inserted into the soil for incubation. The remaining 2 cores were composited and returned to the lab for analysis. Incubation intervals ranged from 50 to 60 days and roughly corresponded to the months of May-June, July-August, September-October, and November-December 1990 (total of 8 months). Samples returned to the laboratory were treated and extracted by the same procedures used for the lab incubations. Ammonium and NO_3^--N production was calculated by subtracting initial nonincubated sample concentrations from incubated sample concentrations. Net N-mineralization estimates were obtained by summing the changes in NH_4^+-N and NO_3^--N over each field-incubation period. For the 5-cm-depth cores, data are reported on an OM basis.

The transplant study was designed with 2 soil types (urban and rural cores) and 2 land-use types (urban and rural land-use environments). We used two-way analysis of variance (ANOVA) to determine the treatment level effects of soil and land-use type and their interactive effect on total accumulations of net inorganic N (NH_4^+-N + NO_3^--N) and NO_3^--N after 4 successive bimonthly field incubations of transplanted cores between the urban and rural land-use types. We used a Least Squares Means test to compare means of 8-month total accumulations of net inorganic N (NH_4^+-N + NO_3^--N) and NO_3^--N between urban and rural soils incubating within or between land-use types.

Surface and subsurface soil field incubations

In addition to the soil cores taken from the 12 stands in 1990, another year of native soil measurements was added in 1993 for the original 12 plots and an additional 5 plots in surface (A and Oe horizons) and subsurface (upper B horizon) layers (figure 2). We differentiated surface and subsurface horizons because the rural and some suburban stands (S1; ML 3 and 4; and MR 4) had a 1- to 2-cm layer of Oe organic material, or mor soil characteristics, while most urban and some suburban stands (S2 and MR3) exhibited typical mull characteristics (Table 1).

Four 5-cm diam cores to a 10-cm depth were taken from each quadrat monthly from April to December 1993 (total of 9 months). Two cores were placed in polyethylene bags, sealed, and inserted into the soil for incubation. The remaining 2 cores were returned to the lab for analysis and treated and extracted by the procedures used for the transplanted cores and lab incubations; however, these 10-cm cores were separated into surface and subsurface material prior to extraction. The surface material for the stands with mull soils was primarily mineral soil, whereas the stands with mor soil characteristics had surface layers of both organic and mineral materials. Calculations of ammonium-N and NO_3^--N production and net N-mineralization and nitrification rates were the same as with the transplanted cores but occurred monthly rather than bimonthly. For the 10-cm-depth cores, data on net N-mineralization and nitrification are reported on an OM basis. Relative nitrification was calculated as the proportion of inorganic N accumulated as nitrate at the end of the field-incubation period for all nitrifying samples. We used a one-way ANOVA to test the difference among means of three land-use types (urban, suburban, and rural). Hochberg's Method for unequal sample sizes (SAS, 1997) was used to test for differences between final means of

nitrogen transformation rates for the land-use and soil types. Where appropriate, data were transformed using a $\log_{10}$ transformation to reduce heteroscedasticity as determined by the Box-Cox transformation tests (Box and Cox, 1982).

Results

Potential rates

The laboratory incubations of native soil suggest that urban nitrification is greater than rural nitrification, with suburban nitrification intermediate and not statistically different from either urban or rural stands (Table 2). Approximately 70% of the inorganic N measured by the end of the incubation period was composed of nitrate in urban and suburban stands compared with 43% in rural stands on both a volumetric and OM basis. For net nitrification, urban stands had approximately a 5-fold greater accumulation of NO_3^--N than rural stands on an OM and volumetric basis. Moreover, results of a one-way ANOVA indicated that by the end of the incubation period, net nitrification differed significantly ($p = 0.04$ and 0.01) between land-use types on both an OM and volumetric basis, respectively (Table 2). The repeated measures MANOVA was statistically significant for the accumulation curves of NO_3^--N on a volumetric basis ($p = 0.05$), but not total inorganic N (NH_4^+-N + NO_3^--N), between land-use types over time (Table 3). The latter result was due in part to the low power of the experimental design. Notwithstanding, soils from urban stands had 38% greater net N-mineralization on an OM basis and 56% greater net N-mineralization on a volumetric basis than soils from rural stands.

Soil core transplants

In the 8-month field-incubation period, the transplant experiment revealed that soil type and to a lesser extent land-use type affected net N-mineralization and nitrification rates (figure 3

Table 2. Potential net N mineralization and nitrification means by land-use type for 8-week laboratory incubations along urban-rural transect in New York City metropolitan area

Property	Land-use types[1]			p value (ANOVA)
	Rural	Suburban	Urban	
Net N				
μg N g^{-1} OM	370[a]	403[a]	551[a]	0.29
μg N cm^{-3}	48[a]	54[a]	77[a]	0.13
Net NO_3^-				
μg NO_3^- g^{-1} OM	75[a]	277[a,b]	395[b]	0.04
μg NO_3^- cm^{-3}	12[a]	32[a,b]	55[b]	0.01

[1]Means with different letters are significantly different at $p > 0.05$; values are the mean of 4 plots per land use type; $n = 4$ samples per plot. Statistical analyses were performed on transformed data ($\log_{10}$).

Table 3. Repeated measures MANOVA of incubation period (time) and land-use type for amount of inorganic N accumulated during 8-week laboratory incubation. Data for total inorganic N accumulated by end of incubation period presented in Table 2

Source	Wilks'λ	*F*	df	*p*
μg N g^{-1}OM				
Time	0.05556	42.5	(2,4)	0.0007
Time*land-use type	0.56982	1.89	(2,4)	0.25
μg N cm^{-3}				
Time	0.05218	45.4	(2,4)	0.0006
Time*land-use type	0.45290	3.02	(2,4)	0.14
μg NO_3^- g^{-1} OM				
Time	0.20701	9.58	(2,4)	0.02
Time*land-use type	0.37846	4.12	(2,4)	0.09
μg NO_3^- cm^{-3}				
Time	0.13967	15.4	(2,4)	0.007
Time*land-use type	0.30556	5.68	(2,4)	0.05

Note: Statistical analyses were performed on transformed data ($\log_{10}$).

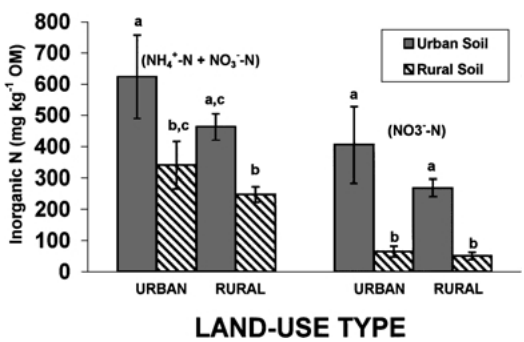

Figure 3. Mean (±S.E.) of total inorganic N (NH_4^+-N + NO_3^--N) and NO_3^--N accumulated after 8-month incubation period in reciprocally transplanted soil cores between urban and rural stands; values are the means of 4 plots per land-use type (32 soil cores per land-use type each sampling period). Results of ANOVA and Least Squares Means tests are presented in Table 4 and in text, respectively. Means with different letters among soil types are significantly different at $p > 0.05$.

and Table 4). While soil and land-use type effects were not independently significant in the two-way ANOVA for net N mineralization and nitrification ($p = 0.14$ and 0.25 for land use type and $p = 0.68$ and 0.35 for soil type, respectively), there were significant interactive effects at the end of the 8-month field-incubation period ($p = 0.009$ and $p = 0.0009$ for net N mineralization and nitrification, respectively). A statistically significant interactive effect suggests that urban and rural soil types responded differently to land use. While mean comparisons of N-transformation rates between urban and rural stands for each soil type

Table 4. Results of a two-way ANOVA of main effects of land-use and soil type for net inorganic N accumulated seasonally and after 8 successive months of field incubations (8-month total data are summarized in figure 3; seasonal data presented in Table 5)

Source	df	*F*	*p*
Total net N (OM)			
Land-use type	1	2.48	0.14
Soil type	1	0.18	0.68
Land-use*soil	1	9.52	0.009
Total net NO_3^- (OM)			
Land-use type	1	1.43	0.25
Soil type	1	0.94	0.35
Land-use*soil	1	19.42	0.0009
Spring net N (OM)			
Land-use type	1	0.66	0.43
Soil type	1	0.65	0.44
Land-use*soil	1	14.74	0.002
Spring net NO_3^- (OM)			
Land-use type	1	0.60	0.45
Soil type	1	0.56	0.47
Land-use*soil	1	36.41	<.0001
Summer net N (OM)			
Land-use type	1	5.13	0.043
Soil type	1	1.11	0.31
Land-use*soil	1	3.57	0.08
Summer net NO_3^- (OM)			
Land-use type	1	2.50	0.14
Soil type	1	0.94	0.96
Land-use*soil	1	3.29	0.095
Autumn net N (OM)			
Land-use type	1	0.59	0.46
Soil type	1	0.03	0.87
Land-use*soil	1	0.08	0.78
Autumn net NO_3^- (OM)			
Land-use type	1	1.94	0.19
Soil type	1	0.37	0.56
Land-use*soil	1	12.55	0.004

Note: Statistical analyses were performed on transformed data ($\log_{10}$).

were not statistically significant (Least Squares Means test, $p = 0.42$ and $p = 0.64$ for net N-mineralization and $p = 0.71$ and $p = 0.55$ for net nitrification, respectively), significant differences were found between urban and rural soils within each land-use type (figure 3). Urban soils nitrified nearly 6.3 and 5.4 times more than rural soils incubating in urban and

Table 5. Means (S.E.) of net inorganic N accumulated in field incubations by land-use type and season. Soil cores were collected in urban and rural forest stands and reciprocally transported to the same urban and rural stands. Values are mean of 4 plots per land-use type; $n = 4$ samples per plot. Total accumulations after 8-month field incubation are presented in figure 3. Results of an ANOVA for each season presented in Table 4

	Land-use type			
	Urban stand		Rural stand	
Property	Urban soil	Rural soil	Urban soil	Rural soil
Spring				
μg N g^{-1} OM	368 (76)	172 (35)	348 (26)	215 (25)
μg NO_3^- g^{-1} OM	228 (78)	26 (4)	195 (42)	43 (10)
Summer				
μg N g^{-1} OM	144 (52)	95 (48)	71 (43)	12 (5)
μg NO_3^- g^{-1} OM	102 (46)	29 (17)	50 (40)	5 (2.5)
Autumn				
μg N g^{-1} OM	111 (39)	74 (39)	44 (12)	19 (5)
μg NO_3^- g^{-1} OM	76 (29)	10 (5)	24 (0.8)	3 (0.5)

rural stands, respectively (Least Squares Means test, $p = 0.003$ and $p = 0.002$). Similarly, in urban stands total accumulation of inorganic N was 83% higher in urban than in rural soils (Least Squares Means test, $p = 0.043$), whereas in rural stands, urban soils mineralized 87% more N than rural soils, however, this difference was not statistically significant (Least Squares Means test, $p = 0.08$). Comparing soils incubating in their native locations, urban soils incubating in urban stands mineralized more than 2.5 times the amount of N than rural soils incubating in the rural sites (Least Squares Means test, $p = 0.019$). By contrast, urban soils incubating in urban stands exhibited a 8-fold increase in nitrification over rural soils incubating in rural stands (Least Squares Means test, $p = 0.008$).

Seasonal net accumulations of inorganic N for spring, summer, and autumn for the most part exhibited a similar pattern to the total accumulations of NH_4^+-N and NO_3^--N at the end of the 8-month field-incubation period (Table 5). The notable exception to the total accumulation pattern of inorganic N (NH_4^+-N + NO_3^--N) occurred for the summer months where a significant land-use type effect was detected (ANOVA, $p = 0.043$; Table 4). Soils incubating in the urban stands accumulated 53.2% more inorganic N than soils in rural stands (Table 5). Otherwise, there were significant interactive effects of soil and land-use type in the two-way ANOVA for NO_3^--N in the spring and autumn, and total accumulation of inorganic N in the spring incubation periods, respectively (Table 4).

Surface and B-horizon field incubations

Urban and suburban land-use types seem to have a positive effect on net rates of N-mineralization and nitrification in both surface and subsurface layers of soil (figures 4 and 5). The surface layer of suburban and urban stands had 3- and -2.3 fold higher net

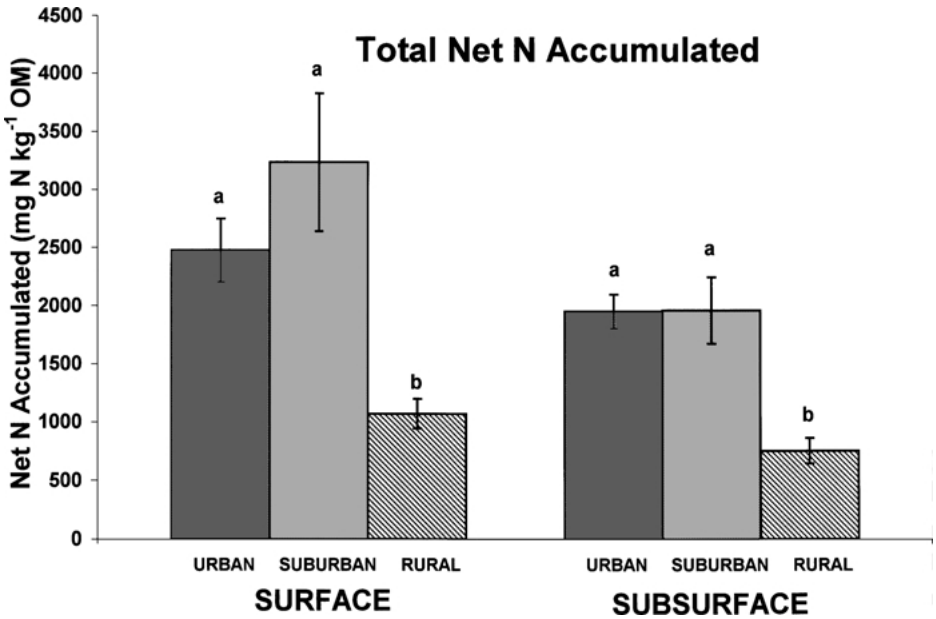

Figure 4. Mean (±S.E.) of total inorganic N (NH_4^+-N + NO_3^--N) accumulated in surface and subsurface soil after a 9-month incubation period in 17 oak forest stands along the urban-rural gradient; values are the mean of 6 urban and suburban plots and 5 rural plots (48 and 40 cores per land-use type per sampling period, respectively). Means with different letters among land-use types are significantly different at $p > 0.05$ for surface and subsurface soils.

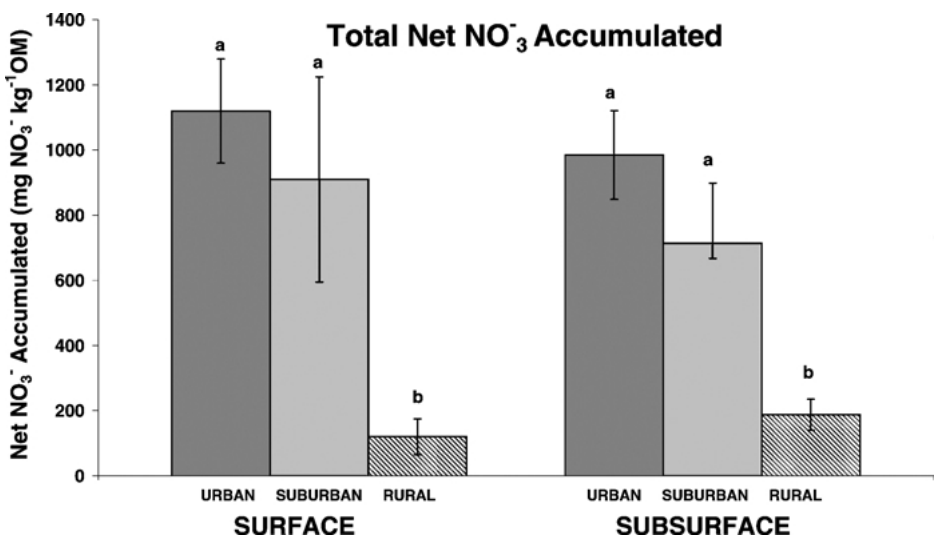

Figure 5. Mean (±S.E.) of total NO_3^--N accumulated in surface and subsurface soil after a 9-month incubation period in 17 oak forest stands along the urban-rural gradient; values are the mean of 6 urban and suburban and 5 rural plots (48 and 40 cores per land use type per sampling period, respectively).

inorganic N accumulated than that of rural stands by the end of the field-incubation period (figure 4). These differences were statistically significant in a one-way ANOVA ($p = 0.05$). The variation of net N-mineralization was highest in suburban stands, where there was roughly an equal proportion of mull and mor soil types (Table 1, figure 4). The subsurface layer of both urban and suburban stands had a 2.6-fold higher net N-mineralization rate than that in rural stands (ANOVA, $p = 0.01$) (figure 4). There were consistent trends between the surface and subsurface layers across all land-use types; the surface layer accumulated 27, 65, and 42% more net inorganic N than the subsurface layer in urban, suburban, and rural stands, respectively. These means, however, were not statistically separable (t-test, $p = 0.085$, $p = 0.12$, and $p = 0.109$, respectively; figure 4).

As with net N-mineralization, urban and suburban land-use types had a strong positive effect on net rates of nitrification in both the surface and subsurface layers (figure 5). The surface layer of the urban and suburban stands had 9.3- and 7.6-fold higher net nitrate accumulated than that in rural stands by the end of the 8-month period (figure 5). These differences were statistically significant in a one-way ANOVA ($p = 0.03$). In the subsurface layer, urban and suburban stands had 5.3- and 3.8-fold higher net nitrification rates than in rural stands (ANOVA, $p = 0.02$; figure 5).

Since all stands on the transect exhibited a net accumulation of nitrate by the end of the field incubation period (figure 5), we calculated the relative nitrification for each sample location. Roughly half the net NH_4^+-N mineralized was nitrified in the urban stands (figure 6). Relative nitrification in the surface layer was 2.3 and 4.9-fold higher for urban stands than for suburban and rural stands, respectively (ANOVA, $p = 0.001$; figure 6). For all land-use types, the subsurface layer had consistently higher mean percentages of NH_4^+-N nitrified than the surface layer, with rural stands exhibiting a statistically significant 2-fold increase in percent of NH_4^+-N nitrified in the subsurface horizon (t-test, $p = 0.05$; figure 6).

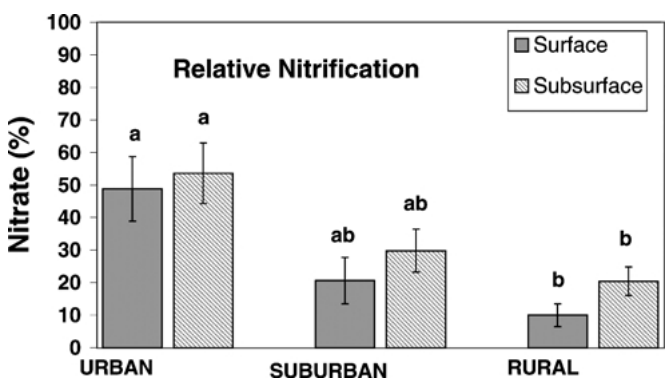

Figure 6. Mean (±S.E.) relative nitrification values in surface and subsurface soil after a 9-month incubation period for 17 oak forest stands along the urban-rural gradient; relative values are the mean from 6 urban and suburban plots and 5 rural plots (48 and 40 cores per land-use per sampling period, respectively).

Discussion

Both laboratory incubation and the buried-bag method test for the quantity and quality of labile C and N in soil (Binkley and Hart, 1989). Buried bags are sensitive to variations in soil temperature while soil moisture content is held essentially constant throughout the incubation period. In this study we report buried-bag data on an OM basis, which corrects for differences in concentration of organic matter among stands. Therefore, differences in net N-mineralization among urban, suburban, and rural forest stands should be primarily due to changes in the quality of soil organic matter and the availability of N in the soil. Soil biota also can strongly affect N-transformation rates (Swift *et al.*, 1979); however, by using buried bags, the effect of larger soil animals such as earthworms will be relegated to their ability to change soil and microbial community structure on a long-term basis.

The relative contribution of site environmental factors and soil origin (soil type) on net N-mineralization and nitrification rates was quantified separately by reciprocally transplanting undisturbed cores between forest stands that are differentially affected by the type of land-use surrounding them. In the transplant experiment, differences in site conditions were investigated by comparing response variables within the same soil placed at both sites; likewise, the potential contribution of differences in soil origin to N-transformation rates can be assessed by comparing the 2 soil types incubating in the same stands. Therefore, reciprocally transplanted buried bags can determine the relative importance of long-term effects (mor vs. mull soil characteristics, quality of soil organic matter, N deposition, and other site history effects) by comparing responses between soil cores of different origins. Short-term effects (soil temperature) are determined by comparing differences between land-types.

The field transplant, potential, and *in situ* measurements of surface and subsurface soil indicated that soil type and the net effects of both sets of factors, resulted in higher net N-mineralization and nitrification rates in urban and suburban stands than in rural stands along this urban-rural gradient (figures 3–5). There also were differences in net nitrification rates between native soil samples in the lab incubations, which controlled for moisture and temperature (Tables 2–3). Determining the specific aspect of the urban and suburban environments that may have caused these changes was not the focus of this study. We use measurements from earlier studies (Pouyat *et al.*, 1994, 1995, 1997), along with site information collected in this study to speculate on specific urban and suburban site factors that affected net N-mineralization and nitrification rates along the gradient.

Short-term effects

Given the low statistical power of the reciprocal transplant experiment, there were no statistically significant ($p > 0.05$) short-term effects (soil temperature) found between the urban and rural stands. The trend for each soil, however, was to have higher mean accumulations of inorganic N in urban vs. rural stands with the exception of nitrification rates in the rural soil type (figure 3). The absence of a short-term effect (soil temperature) was not expected. In previous measurements along the transect, mean soil temperatures (2-cm depth, annual average in 1989) were 2° to 3°C higher in urban than in rural stands due to the urban heat island effect (Pouyat, 1992). This temperature difference has the potential to increase

N-transformation rate by as much as 20% between urban and rural sites, assuming a doubling of rate with an increase in temperature of 10°C (Pouyat *et al.*, 1997). While differences for the urban soil type in net N-mineralization and nitrification between land-use types exceeded 20%, (34.6 and 73.2% for urban soils incubating in urban and rural stands, respectively) the observed means were not separable given the statistical power of the reciprocal transplant experimental design.

Long-term effects

Potential net nitrification rates for soils from urban stands were significantly higher than those for soils from rural stands (Table 2). Consistent with lab incubation results, native soil cores incubated in urban and suburban stands had higher net N-mineralization rates in surface and subsurface soil than those incubated in rural stands (figures 4 and 5). Results for the reciprocal soil-core transplants also suggest that the urban soils mineralize and nitrify N at a higher rate than rural soils along this urban-rural gradient (figure 3). Results of these lab- and field-incubation measurements suggest that the relatively higher net N-mineralization rate is a result of a larger pool of mineralizable N in the surface 10 cm of soil in the urban and suburban stands. These larger pools may be the result of differences in the quality of soil organic matter, earthworm abundance and activity, and N deposition rates that have been measured along this urban-rural land-use gradient (Groffman *et al.*, 1995; Steinberg *et al.*, 1997; Lovett *et al.*, 2000).

High earthworm activity can increase mineralizable N pools in soil and may explain differences among urban, suburban, and rural stands. Earthworms ingest a large proportion of the total organic matter in surface soils and through fragmenting and cast-forming activities stimulate microbial activity, increasing N availability (Scheu, 1987; Blair *et al.*, 1995). Moreover, earthworm activity has created a mull soil condition in the urban and in some suburban stands, and it is in these stands that N-transformation rates seems the highest (Pouyat, 1992; Zhu and Carreiro, 1999; Table 1). In mull soils, earthworms could alter N-mineralization and nitrification rates by disturbing resident microbial populations, particularly fungi, and producing casts, which have more favorable chemical and physical properties for microbial activity than bulk mineral soil (Blair *et al.*, 1995; Zhu and Carreiro, 1999). In fact, while net N-transformation rates were higher in urban and suburban stands in this and other studies (Pouyat *et al.*, 1997; Zhu and Carreiro, 1999), investigations of soil biota have reported lower litter fungal abundances in urban than in rural stands (Pouyat *et al.*, 1994). The exact relationship between earthworm and other soil-organism distributions and net N-mineralization and nitrification rates along this land-use gradient needs further study.

The quality of organic material in the surface 10 cm of soil also may be a factor affecting N-mineralization rates. In previous studies investigating the stands sampled in this study, decay rates measured in lab and field incubations demonstrated that leaf litter of red oak collected from rural stands decomposed and mineralized N more rapidly than suburban and urban red oak leaves (Carreiro *et al.*, 1999; Pouyat and Carreiro, 2003). Typically, poor litter-quality input either reduces the rate at which labile C mineralizes N or increases the amount of organic matter transferred to recalcitrant pools, or both (Lamb, 1975). In fact, preliminary measurements of soil C pools along the urban-rural transect suggest that

recalcitrant C pools are higher in urban forest soils relative to rural forest soils (Groffman *et al.*, 1995). These results suggest that while urban site factors such as high N deposition, earthworm abundances, and soil temperature tend to accelerate decay and potential N mineralization rates, urban litter and quality of soil organic matter may tend to reduce decay rates and subsequently lower N availability in these soils. In both this study and a previous investigation (Pouyat *et al.*, 1997), there were no statistically significant relationships between net N-mineralization rate and the C:N ratios of the forest floor or soil even though our data are expressed on an OM basis. A possible explanation is that any increase in available soil N, whether due to earthworm activity or exogenous inputs of N, can result in more rapid and efficient utilization of litter by decomposers (Chapman *et al.*, 1988) even if litter quality is more recalcitrant (Seastedt, 1984).

In the transplant experiment, differences between urban and rural site effects for accumulations of NO_3^--N were not apparent for rural soils (figure 3). Moreover, in urban and suburban sites, nitrification accounted for up to 50% of the total N mineralized in urban soil cores compared to 20% for rural cores, regardless of location (figure 3). Results were similar in native soil-core incubations for both surface and subsurface soils along the transect (figure 6). The relatively high nitrification rates and high proportions of mineral N accumulated as nitrate in the urban and some suburban stands are consistent with lab incubation results obtained by Zhu and Carreiro (1999) who found that urban and suburban soils collected along the transect nitrified rapidly. They also showed that the nitrification occurring in these soils was by chemoautotrophic rather than heterotrophic microorganisms.

That net nitrification rates are higher in the urban and some suburban plots than in the rural stands can be attributed to various factors. It is generally believed that NH_4^+-N availability can control rates of autotrophic nitrification (Vitousek *et al.*, 1982). However, the results of our reciprocal soil-core transplants support this hypothesis only partially. While there was a strong positive relationship between net N-mineralization and nitrification rates in urban and to a lesser extent suburban soil cores, there was no observable relationship with the rural cores (figures 3 and 6). The overall lack of net rates of nitrification in rural stands in this study also was found in a lab microcosm study by Steinberg *et al.* (1997). They reported that rural soils had higher net inorganic N accumulations than urban soils, but that there was no apparent accumulation of NO_3^--N even when earthworms were added to some microcosms. The authors suggested that long-term effects of site history, such as atmospheric deposition of N and the promotion of mor-to-mull shift by earthworm activity in urban and some suburban stands, may favor nitrifying bacteria in their competitive interactions with heterotrophic organisms (particularly fungi) for NH_4^+-N (Steinberg *et al.*, 1997).

We conclude that along this urban-rural gradient, soils in oak-dominated stands exhibit higher net nitrification and, to a lesser degree, net N-mineralization rates in urban and suburban stands than in rural stands. Results from lab and field incubations indicate that long-term effects play a major role in this response along the urban-rural gradient. By contrast, soil temperature, as affected by the urban heat-island effect, does not appear to be a major short-term factor affecting differences in N-transformation rates along this gradient, at least for the field incubation period of this study. Possible long-term effects include shifts in the structure of forest soil organic horizons (mor to mull) by nonnative earthworms,

chronic N deposition, or another site history effect we have not accounted for along this urban-rural transect. The higher net nitrification rates in urban and suburban forest stands suggest that these forests have the potential to loose high amounts of N through soil leaching. Future research is needed to experimentally investigate the effect of individual urban factors on N-transformation rates in forest soils along this urban-rural transect.

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