

Short communication

## Soil nitrogen cycling under litter and coarse woody debris in a mixed forest in New York State

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### Abstract

Coarse woody debris (CWD) could alter N availability and transformations in the underlying soil and therefore contribute to spatial heterogeneity and influence ecosystem loss of N. We measured soil N concentrations and transformations in soil beneath CWD and beneath a litter layer at a mixed forest in NY State. We found that total and microbial biomass N was lower and that microbial biomass C-to-N ratio was higher in soil beneath CWD. Rates of N<sub>2</sub>O production and denitrification enzyme activity were reduced beneath CWD. These results suggest that CWD is an important controller of spatial heterogeneity in N dynamics and may influence the magnitude of N loss in temperate forests. © 2005 Elsevier Ltd. All rights reserved.

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The contribution of coarse woody debris (CWD) to N cycling has been studied by measuring net changes in CWD N over time or in chronosequences, and through measurements of specific transfers, e.g. N<sub>2</sub> fixation or leaching (e.g. Alban and Pastor, 1993; Krakina et al., 1999; Spears et al., 2003; Brunner and Kimmins, 2003). However, CWD could alter N availability and transformations in the underlying soil, and therefore contribute to spatial heterogeneity and influence ecosystem losses. In this study we report differences between soil under CWD and soil under a litter layer at a mixed forest in NY State.

Coarse woody debris has the potential to influence soil N cycling through several mechanisms, including: leaching dissolved N and dissolved organic C (DOC) (Yavitt and Fahey, 1985; Mattson et al., 1987; Spears et al., 2003; Hafner et al., 2005), the latter of which may lead to N immobilization (Magill and Aber, 2000; Fierer et al., 2001); preventing or promoting the accumulation of leaf litter, which has a higher N concentration than wood (Daubenmire

and Prusso, 1963; Cowling and Merrill, 1966; Lang and Forman, 1978; Yavitt and Fahey, 1986); translocation of N by decomposer fungi (Boddy and Watkinson, 1994); and alteration of physical properties that affect microbial activity, e.g. moisture effects on denitrification (Groffman and Tiedje, 1989).

There have been few assessments of the effects of CWD on soil N cycling. Busse (1994) reported lower inorganic N beneath logs compared to soil without logs in a lodgepole pine forest, and Spears et al. (2003) reported lower gross N mineralization beneath logs compared to soil beneath the forest floor in a Douglas-fir forest. There have been no measurements of the effects of CWD on gaseous N production in soil. In this study, our objective was to determine if soil N concentrations and transformations beneath CWD differed from those beneath litter in a temperate mixed forest.

The study site was a ca. 30 y old lowland forest in southeastern New York State (see Hafner et al., 2005). The site existed as a pasture with some trees until 1972, and the age of trees is therefore variable. The 7.8-ha site is flat (elevation range 91–94 m), primarily with mineral soils, some areas of which are regularly flooded. The litter layer is thin (ca. 10 mm) and is the only organic horizon over most of the site, probably due to flooding. Annual precipitation is 1070 mm and temperature ranges from –2.7 °C in January

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to 21.8 °C in July (monthly means, 1988–2002) (Kelly, 1999, 2002). Dominant tree species include red maple (*Acer rubrum* L.), sugar maple (*Acer saccharum* Marshall), white pine (*Pinus strobus* L.) and American elm (*Ulmus americana* L.).

This study employed a blocked design. At each of six blocks, a CWD plot (the soil beneath a log), and a 4×4 m litter plot were established. CWD plots were selected from a census of all downed logs, stratified by decay class, and two logs from each of three decay classes were included (dia 17–35 cm at center) (see Hafner et al., 2005). Litter plot location was selected randomly from two locations 1 m from the log edge.

Duplicate 4.7-cm dia soil samples were collected in summer (late July) and fall (late September–early October), 2002, to 10 cm, sieved (6.4 mm), and mixed. Subsamples (10 g) were dried at 105 °C to determine moisture content. Extractable  $\text{NO}_3^-$  and  $\text{NH}_4^+$  were determined following Robertson et al. (1999). For analysis of pH, organic C, and total Kjeldahl N, dried and sieved (2 mm) samples from summer and fall were mixed equally by mass. Soil pH was measured in a 1:2 suspension (soil:DDW) (Bickelhaupt et al., 1983). Organic C was determined using the Walkley–Black wet oxidation method, and N was determined using a macro-Kjeldahl method (Bickelhaupt et al., 1983).

For the measurement of respiration and  $\text{N}_2\text{O}$  production, two intact soil cores (4.7-cm dia) were collected from each litter and CWD plot using a steel corer. Samples from the same plot were incubated together in a sealed chamber in the laboratory, and the concentrations of  $\text{O}_2$  and  $\text{N}_2\text{O}$  were measured at two points over a 3-h incubation via gas chromatography. Respiration was expressed in units of  $\text{CO}_2\text{-C}$ . Rates from the summer and fall were averaged for the statistical analyses.

During the summer sampling period, mixed, sieved (6.4 mm) samples were analyzed for microbial properties. The chloroform fumigation-incubation method was used to estimate microbial biomass C and N, and potential net N mineralization and nitrification were determined from unfumigated controls, in 10 day laboratory incubations following Groffman et al. (1996). Denitrification enzyme activity (DEA) was determined following Groffman et al. (1999).

Data were analyzed to determine if differences existed between measured variables under the two conditions via the Wilcoxon paired-sample test (Zar, 1999), using SAS (v. 9, SAS Institute, Cary, NC), with  $\alpha=0.10$ .

The only chemical property to show a significant difference beneath CWD and litter was total Kjeldahl N, which was slightly lower under CWD than under litter (Table 1). Nitrate was detected in only two plots; therefore, it was grouped with  $\text{NH}_4^+$ . The mean concentration of inorganic N was much lower beneath CWD than litter, but the difference was not statistically significant ( $P=0.22$ ), probably due to a small sample size and resulting low power of the analysis.

Mean respiration rates were very similar between conditions (Table 2). However, the production of nitrous oxide was much lower under CWD than with litter (Table 2). Microbial biomass N was lower, and microbial biomass C-to-N ratio higher, beneath CWD (Table 2). Denitrification enzyme activity under CWD was ca. one-half that under litter (Table 2). Though both potential net N mineralization and potential net nitrification showed much lower rates under CWD, differences were not statistically significant, probably due to the small sample size and resulting low power of the analysis (Table 2).

The reduction in N concentrations and transformation rates beneath CWD could be caused by any of the mechanisms discussed above that reduce N availability in the underlying soil. At this site, litter has not accumulated on top of or adjacent to CWD, and, while concentrations of inorganic N are similar in litter and CWD leachate, reduced leachate fluxes probably lead to reduced leaching of inorganic N from CWD (Hafner et al., 2005). Conversely, concentrations of DOC in CWD leachate are ca. nine-fold the concentrations in litter leachate (Hafner et al., 2005), possibly contributing to N limitation of soil microbes, and reduced rates of N transformations. Fungal translocation of N from the soil into CWD probably plays an important role as well. In other forested ecosystems, decaying CWD has been shown to accumulate applied N or N from the soil (Vitousek and Matson, 1984; Zimmerman et al., 1995). At this site, the total N content of the six sampled logs is greater than the difference in N content in soil beneath CWD and litter (to the sampled depth of 10 cm), demonstrating that N

Table 1

Means of soil physical and chemical properties, and results of the Wilcoxon paired-sample test<sup>a</sup>

Variable	Mean		Standard error		P
	CWD	Litter	CWD	Litter	
Moisture (%mass)	28.4	26.7	2.96	1.99	0.31
Bulk density ( $\text{Mg m}^{-3}$ )	0.808	0.839	0.0342	0.0148	0.31
Extractable $\text{NO}_3^- + \text{NH}_4^+$ ( $\text{mg N kg}^{-1}$ )	2.20	3.56	0.319	0.812	0.22
$\text{H}^+$ activity ( $\mu\text{M}$ )	21.5	19.5	8.76	5.10	0.44
Total Kjeldahl N (%)	0.301	0.335	0.00858	0.00695	0.031
Organic C (%)	4.94	4.91	0.443	0.218	0.84

<sup>a</sup> For all variables  $n=6$ .

Table 2  
Means of microbial variables, and results of the Wilcoxon paired-sample test<sup>a</sup>

Variable	Mean		Standard error		P
	CWD	Litter	CWD	Litter	
Respiration (mg C kg <sup>-1</sup> d <sup>-1</sup> )	57.6	55.2	6.80	2.71	0.69
N <sub>2</sub> O production (μg N kg <sup>-1</sup> d <sup>-1</sup> )	2.21	4.84	0.628	0.808	0.031
Microbial biomass C (mg C kg <sup>-1</sup> )	1110	1040	115	102	0.84
Microbial biomass N (mg N kg <sup>-1</sup> )	77.5	90.6	6.53	6.03	0.063
Microbial C-to-N ratio (g g <sup>-1</sup> )	15.2	12.0	2.35	1.57	0.063
Potential net N mineralization (μg N kg <sup>-1</sup> d <sup>-1</sup> )	474	1110	88.2	412	0.16
Potential net nitrification (μg N kg <sup>-1</sup> d <sup>-1</sup> )	65.7	120	32.1	52.9	0.25
DEA (mg N kg <sup>-1</sup> d <sup>-1</sup> )	11.3	18.1	2.88	4.92	0.063

<sup>a</sup> For all variables  $n=6$ .

accumulation in CWD could be of a sufficient magnitude to account for the lower N content of soil beneath it.

It is probable that a reduction in N availability is responsible for the decreased microbial biomass N, and reduced N<sub>2</sub>O production and DEA. These results suggest that CWD increases the spatial heterogeneity of N cycling in forest soils, creating areas of low N availability and N limitation of microbes. This variability may influence the establishment and persistence of vegetation with different N requirements, and could play a role in determining species diversity in forested ecosystems. Additionally, sampling schemes should consider this source of spatial heterogeneity.

While it is important to note that we did not measure rates of gaseous N loss in the field, our results suggest that CWD may be an important regulator of denitrification and N<sub>2</sub>O flux in these forests. Additionally, reductions in N concentrations could cause a reduction in N leaching. Though the greatest effects would occur on a microsite scale (e.g. directly beneath CWD), reductions in N losses may be important on an ecosystem scale. Assuming background rates of gaseous N loss of 30 kg ha<sup>-1</sup> y<sup>-1</sup> for a poorly drained forest (Groffman and Tiedje, 1989), CWD cover of 5%, and a 75% reduction in gaseous N loss under CWD, the effect of CWD on net N loss due to a reduction in gaseous N loss would be 1 kg N ha<sup>-1</sup> y<sup>-1</sup>. This value is greater than most estimates of the ecosystem contribution of N fixation in CWD (Jurgensen et al., 1984; Brunner and Kimmins, 2003). Effects could be much greater in ecosystems where the cover of CWD is higher. For example, if the same relationships were to hold in old-growth forests in the Pacific Northwest of the US, where cover of CWD may exceed 25% (Harmon and Hua, 1991; Means et al., 1992), CWD could account for a >15% reduction in gaseous N losses from the soil.

Our results suggest that CWD may be an important regulator of N availability and loss in forests on multiple scales. Forests in the northeastern US are aging and exhibiting complex N dynamics (Goodale et al., 2003). As these and other forests age, pools of CWD are increasing, and effects on soil N cycling may become increasingly important. The effects of CWD on soil N cycling should be

considered in evaluations of N dynamics in forested ecosystems.

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