

Provided for non-commercial research and education use.
Not for reproduction, distribution or commercial use.



This article appeared in a journal published by Elsevier. The attached copy is furnished to the author for internal non-commercial research and education use, including for instruction at the authors institution and sharing with colleagues.

Other uses, including reproduction and distribution, or selling or licensing copies, or posting to personal, institutional or third party websites are prohibited.

In most cases authors are permitted to post their version of the article (e.g. in Word or Tex form) to their personal website or institutional repository. Authors requiring further information regarding Elsevier's archiving and manuscript policies are encouraged to visit:

<http://www.elsevier.com/copyright>



Nitrogen immobilization by wood-chip application: Protecting water quality in a northern hardwood forest

Peter M. Homyak^{a,*}, Ruth D. Yanai^b, Douglas A. Burns^c,
Russell D. Briggs^b, René H. Germain^b

^a *Department of Environmental Sciences, University of California, Riverside, 900 University Avenue, Riverside, CA 92521, United States*

^b *SUNY College of Environmental Science & Forestry, 1 Forestry Dr., Syracuse, NY 13210, United States*

^c *US Geological Survey, 425 Jordan Road, Troy, NY 12180, United States*

Received 28 July 2007; received in revised form 28 December 2007; accepted 9 January 2008

Abstract

Forest harvesting disrupts the nitrogen cycle, which may affect stream water quality by increasing nitrate concentrations, reducing pH and acid neutralizing capacity, and mobilizing aluminum and base cations. We tested the application of wood chips derived from logging slash to increase immobilization of N after harvesting, which should reduce nitrate flux to streams. In August 2004, a stand of northern hardwoods was patch-clearcut in the Catskill Mountains, NY, and four replicates of three treatments were implemented in five 0.2-ha cut patches. Wood chips were applied to the soil surface at a rate equivalent to the amount of slash smaller than eight inches in diameter (1× treatment). A second treatment doubled that rate (2×), and a third treatment received no chips (0×). Additionally, three uncut reference plots were established in nearby forested areas. Ion exchange resin bags and soil KCl-extractions were used to monitor nitrate availability in the upper 5–10 cm of soil approximately every seven weeks, except in winter. Resin bags indicated that the wood chips retained 30% or 42% of the nitrate pulse, while for KCl extracts, the retention rate was 78% or 100% of the difference between 0× and uncut plots. During the fall following harvest, wood-chip treated plots had resin bag soil nitrate concentrations about 25% of those in 0× plots ($p = 0.0001$). In the first growing season after the cut, nitrate concentrations in wood-chip treated plots for KCl extracts were 13% of those in 0× treatments ($p = 0.03$) in May and about half those in 0× treatments ($p = 0.01$) in July for resin bags. During spring snowmelt, however, nitrate concentrations were high and indistinguishable among treatments, including the uncut reference plots for resin bags and below detection limit for KCl extracts. Wood chips incubated in litterbags had an initial C:N of 125:1, which then decreased to 70:1 after one year of field incubation. These changes in C:N values indicate that the wood-chip application can potentially immobilize between 19 and 38 kg N ha⁻¹ in the first year after harvesting, depending on the rate of wood-chip application. Our results suggest that the application of wood chips following harvesting operations can contribute to the protection of water quality and warrant additional research as a new Best Management Practice following cutting in regions that receive elevated levels of atmospheric N deposition.

© 2008 Elsevier B.V. All rights reserved.

Keywords: Water quality; Wood chips; C:N ratio; Nutrient cycling; Forest harvesting; Northern hardwood forest

1. Introduction

Intensive harvesting disrupts the nitrogen (N) cycle of forested ecosystems (Hendrickson et al., 1989; Aber et al., 2002), which can have negative impacts on stream water quality through reduced plant nutrient uptake and a resulting increase in the export of nitrate to streams (Hornbeck et al., 1986; Burns and Murdoch, 2005). Because nitrification is an acidifying process (Driscoll and Schaefer, 1989; Murdoch and Stoddard,

1992), intensive harvesting can cause other changes in water quality such as a lowering of pH and acid neutralizing capacity (ANC) and mobilization of aluminum and base cations from forest soils (Hornbeck et al., 1986; Martin et al., 2000; Burns and Murdoch, 2005).

Changes in stream chemistry after forest harvesting are of special concern in areas with elevated N deposition (Fenn et al., 1998), such as the Catskill Mountains in New York (NY), where N in wet deposition is among the highest in the US (6–7 kg N ha⁻¹ year⁻¹) (NADP, 2006). Watersheds in this region are particularly sensitive to forestry operations; harvesting has been shown to induce high rates of nitrate leaching (Burns and Murdoch, 2005; Wang et al., 2006). In

* Corresponding author. Tel.: +1 951 827 3404.

E-mail address: peter.homyak@email.ucr.edu (P.M. Homyak).

northern hardwoods, such water-chemistry changes are typically greatest during the first two years after forest harvesting (Hornbeck and Kropelin, 1982; Hornbeck et al., 1986; Martin et al., 2000).

Previous studies have documented the effects of forest harvesting in areas of elevated N deposition (Hornbeck and Kropelin, 1982; Hornbeck et al., 1986; Dahlgren and Driscoll, 1994). Following a 24-ha clearcut in the Catskill Mountains, stream water nitrate concentrations rose to $1400 \mu\text{mol L}^{-1}$ from background levels of $20 \mu\text{mol L}^{-1}$ during the first year after cutting (Burns and Murdoch, 2005). Not only did nitrate concentrations rise above EPA water quality standards of $710 \mu\text{mol L}^{-1}$ (EPA, 2002), but inorganic monomeric aluminum was mobilized from soils into stream water where it reached toxic levels and caused the death of brook trout held in bioassay cages (Baldigo et al., 2005). Likewise, at a nearby shelterwood cut, adverse changes in stream chemistry were detected despite the removal of only 33% of tree basal area (Wang et al., 2006). In this case, inorganic monomeric aluminum concentrations also rose briefly to toxic levels after the cut. Clearly, the design and implementation of post-harvest land management practices designed to minimize adverse water-chemistry changes are warranted.

Best Management Practices (BMPs) have been used to minimize the negative effects of harvesting on forested ecosystems and water resources (Martin and Hornbeck, 1994; Nyland, 2002). These practices have addressed water quality primarily by minimizing the export of sediments to streams through the protection of stream crossings, riparian areas, and forest soils (Martin, 1988). None of the well known and frequently practiced BMPs directly addresses the disruption of nutrient cycling that results in changes in nitrate, pH, ANC, and inorganic monomeric aluminum concentrations in streams.

Nitrate concentrations in forested streams in the Catskill Mountains generally range from 5 to $40 \mu\text{mol L}^{-1}$ during the summer, but can increase to $100 \mu\text{mol L}^{-1}$ or greater during snowmelt (Murdoch and Stoddard, 1992; Lovett et al., 2000). However, every October, coincident with leaf fall, stream water nitrate concentrations typically fall below detection ($\sim 1 \mu\text{mol L}^{-1}$) (Murdoch and Stoddard, 1993). Dissolved organic carbon freshly derived from leaves is readily available to microbial communities and associated with microbial immobilization of nitrate (Sobczak et al., 2003). These observations suggest that a practice that increases the immobilization of N in the forest floor by adding material high in C:N after logging may reduce nitrate flux to streams and limit associated adverse water chemistry changes.

Many studies support the notion that high soil C:N ratios are correlated with low rates of nitrification and reduced N losses. At the Hubbard Brook Experimental Forest in New Hampshire, high C:N ratios in soil organic matter were inversely correlated with net nitrification rates, suggesting that nitrification was limited by low ammonium (NH_4^+) supplies or by factors such as low pH or tannins associated with a high C:N (Venterea et al., 2003). In another study in a Mediterranean shrubland, wood shavings mixed with the litter layer decreased the availability of mineral N in the soil, suggesting an increase in N

immobilization (Gallardo and Merino, 1998). In soils across a 100-year chronosequence of upland hardwood stands in Indiana, a high C:N ratio of woody debris was associated with increased rates of N immobilization (Idol et al., 2003). In a conifer plantation in Japan, sites with added slash had lower nitrate concentrations in soils than sites where slash was removed (Takahashi and Ikeda, 1994). Across New Hampshire, stream nitrate export was inversely related to the ratio of dissolved organic carbon to dissolved organic nitrogen (Goodale et al., 2000). Lastly, a study in the central Appalachians showed how two adjacent watersheds with similar N inputs differed in nitrate outputs because of differences in soil C:N ratios; the watershed with the lower C:N ratio exported more nitrate (Christ et al., 2002).

In this study, we used wood chips derived from logging slash to test whether the addition of organic material high in C:N to forest soils after harvesting would have potential as a new forestry BMP. We hypothesized that wood-chip treated sites would have lower nitrate and ammonium concentrations in the upper 5–10 cm of soil than those receiving no chips, in the year following forest harvest. To measure the success of the wood chips at reducing nitrate and ammonium, we used two levels of wood-chip application and compared them to sites with no chips and uncut reference sites. We monitored nitrate and ammonium with ion-exchange resin bags and soil-KCl extractions. Additionally, we monitored wood-chip decomposition rates with litterbags as a separate estimate of the potential of this material to immobilize N. We also measured nitrification rates in the wood chip mix in laboratory incubations. Finally, we measured soil temperature and moisture in treatment plots to assess whether these environmental variables explained changes in net N mineralization.

2. Materials and methods

2.1. Site description

The study site is located at the Frost Valley YMCA Model Forest in the Catskill Mountains, NY (Fig. 1). The Model Forest lies in the Neversink River basin, which is part of the New York City Catskill/Delaware water supply system. The research site is dominated by a northern hardwood forest with a basal area (BA) of $30.5 \text{ m}^2 \text{ ha}^{-1}$ and relative density of 107%. The study site is predominantly composed of yellow birch (*Betula alleghaniensis* Britton.; BA = $16.3 \text{ m}^2 \text{ ha}^{-1}$), sugar maple (*Acer saccharum* L.; BA = $6.9 \text{ m}^2 \text{ ha}^{-1}$), red maple (*Acer rubrum* L.; BA = $2.0 \text{ m}^2 \text{ ha}^{-1}$), and American beech (*Fagus grandifolia* Ehrh.; BA = $1.0 \text{ m}^2 \text{ ha}^{-1}$). The mean annual temperature at the Slide Mountain weather station in Ulster County, NY, approximately 2 km from our site, is 4.9°C with an annual precipitation of 1565 mm (NRCC, 2005), of which 20–25% falls as snow. Average yearly runoff is estimated at 970 mm, of which 32% occurs during late winter – early spring (March–April), and only 28% occurs during May through September (Firda et al., 1995).

Soils are mapped as bouldery silt loams in the Arnot–Oquaga–Lackwanna association. These soils are lithic and

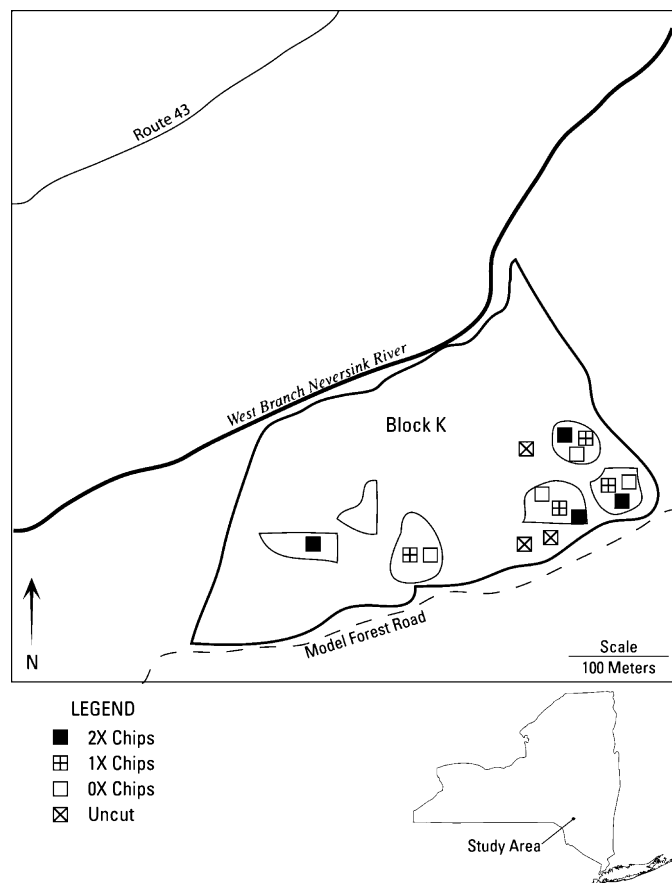


Fig. 1. Map illustrating distribution of wood-chip treatment plots in a patch cut at Frost Valley YMCA Model Forest. Uncut reference plots were located in uncut areas in between cut patches.

typic dystrodepts and fragiudepts with shallow organic horizons (Torns, 1979). The following parameters were measured before forest harvesting from soil samples collected 5–10 cm below the soil surface: pH of 3.7 (standard deviation (S.D.) ± 0.1), loss on ignition of 8.7% (S.D. $\pm 5.0\%$), KCl exchangeable acidity of 2.5 $\text{cmol}_c \text{kg}^{-1}$ (S.D. ± 1.0), cation exchange capacity of 19 $\text{cmol}_c \text{kg}^{-1}$ (S.D. ± 4.7), KCl-extractable nitrate of 2.6 $\mu\text{g g soil}^{-1}$ (S.D. ± 1.1), and KCl-extractable ammonium of 5.6 $\mu\text{g g soil}^{-1}$ (S.D. ± 2.9) (Homyak, 2006).

2.2. Forest harvest

Six patches approximately 0.1 ha each were harvested during August 2004 (Fig. 1). Of the six cut patches, five were used for experimentation; one patch was too small and rocky to accommodate our experimental plots. Loggers removed all merchantable wood, while chipping branches, treetops, and logs smaller than 20 cm in diameter with a commercial wood chipper. The chipped slash was piled near each of the five patches. We used the volume of a cone to estimate the amount of wood chips available for application. Using the weight of a 5-gal bucket filled with the wood-chip mix, we estimated the total mass of each pile. Based on these calculations, we estimated that spreading all the wood chips on the patches would be

equivalent to an application rate of 12 Mg ha^{-1} . These estimates are lower than measurements of coarse woody debris after patch cutting in the Catskill region, which report values of 48 Mg ha^{-1} (Park et al., 2006). It is important to note, however, that coarse woody debris measurements include logs that are greater than 20 cm in diameter, which were removed from the site and not chipped in our study.

From 10 samples collected at each of the five wood-chip piles, we estimated that the average (\pm S.D.) wood-chip size was 34 (± 27) mm in length by 11 (± 6) mm wide and 3 (± 2) mm thick. Chipping also produced sawdust and allowed small twigs and leaves to go through intact. We characterized the wood-chip mixture by subsampling a bucketful from each of the five wood-chip piles. We separated this sub-sample into four categories, and found that 84% by weight was wood chips, 11% was a mixture of sawdust and small particles, 3% was green leaves, and 3% was twigs. Carbon-to-nitrogen ratios for the wood-chip mix were 227:1 for wood chips, 67:1 for mixture of sawdust and small particles, 21:1 for green leaves, and 81:1 for twigs. Based on the weight and C:N ratio of each of the four categories we calculated an average C:N ratio of 200:1 for the wood-chip mix. However, when we analyzed 15 samples from a mixture containing equal amounts of chips from all piles, without separating the sample into the above four categories, we found that the C:N ratio for the mix was 125 (± 13):1. This suggests that although the amount of C in the wood-chip mix was variable, the C:N ratio was generally above 100:1.

To test whether the chips could release N during the initial stages of decomposition, we measured the net nitrification potential of the wood-chip mix (Binkley and Hart, 1989). Nine samples of the mix were incubated in the lab at 22 °C. After 7 days, nitrate concentrations were measured in three of the samples. The remaining samples were measured after 17 and 29 days (triplicate measurements). Samples were shaken in 40 ml of deionized water for 30 min and filtered through 0.7- μm glass microfiber filter paper prior to analysis of nitrate by ion chromatography.

2.3. Treatments

Three treatments were used to test the potential of wood chips to immobilize N. Two of the three treatments received wood chips at two rates, 1 \times and 2 \times . Treatment 1 \times represented the application of wood chips at the rate produced from available slash at each patch (12 Mg ha^{-1}), while 2 \times doubled that amount (24 Mg ha^{-1}). A third treatment, 0 \times , did not receive chipped slash. Three uncut plots were used as reference sites. All treatment and reference plots were monitored from August 2004 through October 2005. Measurements were not taken during the dormant season from November 5, 2004 to April 4, 2005.

Three reference plots were laid out in uncut areas between the cut patches (Fig. 1). At three of the patches, three plots 9.2 m \times 9.2 m were randomly allocated to treatments 0 \times , 1 \times , and 2 \times . The remaining two patches shared the wood-chip treatments by having 2 \times in one patch and 0 \times and 1 \times in the other. The unbalanced arrangement of these plots proved

necessary because space limitations prohibited the placement of all treatments at each cut patch.

Wood chips were distributed by hand on the forest floor at each of the treated plots. The wood-chip application in 1× treated plots did not completely cover the forest floor, while the application rate in 2× plots covered most of the forest floor.

2.4. Nitrogen measurements

Ion exchange resin bags were used to monitor N dynamics in experimental plots. This technique uses a mixture of cation and anion exchange resins (J.T. Baker, IONAC NM-60H⁺/OH⁻ beads) to which nitrate and ammonium adsorb. The resin bags collect nitrate and ammonium from water that infiltrates through the soil during and after rain events (Wyland and Jackson, 1993). The resin bags were constructed following the methods adopted by Burns and Murdoch (2005), which were slightly modified from Binkley and Matson (1983). Prior to each use, the ion-exchange resin was placed in 10% HCl for one hour and then rinsed with deionized water. After the rinse, approximately 6 g (dry weight) of the resin were placed inside a nylon stocking that contained a ring made by cutting 1 cm slices of a 5-cm diameter PVC tube. The purpose of the ring was to provide rigid support, to maintain a constant diameter opening for water to infiltrate through the resin, and to facilitate burying of the resin bags in soils.

On August 19, October 9, and November 27, 2004, and April 9, May 30, July 17, and September 5, 2005, three ion exchange resin bags were placed in each of the 15 treatment plots (Fig. 1). These were incubated in the field for approximately 7 weeks (except during winter) and collected on October 9 and November 27, 2004, and April 9, May 30, July 17, September 5, and October 23, 2005. To avoid edge effects, we monitored N dynamics within the inner 4 m² at each of the plots, leaving a buffer of at least 3.5 m.

To place the resin bags, a spade was inserted to an approximate depth of 5 cm and a bag was slipped beneath the blade. This process was carried out without disturbing the soil upslope of the newly buried resin bag so the same area could be used for future sampling inside the plot. Because the site is characterized by shallow organic horizons (approximately 2 cm thick), resin bags collected water infiltrating through the organic layer and the upper part of the mineral soil. Resin bags were collected from the field at the end of each incubation period and kept cold in a container filled with ice. Samples were taken to the lab and processed within 48 h for extraction of nitrate and ammonium with 50 ml of 2 M KCl. Samples were then filtered through 0.7- μ m glass microfibre filter paper and a 30-ml aliquot sample was frozen for later analysis of nitrate and ammonium by a colorimetric method (Eaton et al., 1995) with the use of a Bran + Luebbe Autoanalyzer 3.

In addition to the ion exchange resin bags, on October 9 and November 27, 2004, and April 9, May 30, July 17, September 5, and October 23, 2005, a soil sample was collected at each of the 15 plots (Fig. 1) for analysis of KCl-extractable nitrate and ammonium. Soil-KCl extractions indicate N availability in the soil at the time of sampling. Soil extractions were conducted in

the field by thoroughly mixing a soil sample (about 200 g) at each of the 15 plots from the upper 5–10 cm of soil and shaking a sub-sample from this mix (approximately 10 g) with 30 ml of 2 M KCl. The dry weight of the sub-sample mixed with the KCl solution was estimated by drying another field-moist sub-sample in an oven at 104 °C to constant weight (approximately 48 h). In the lab, the KCl extracts were filtered through 2.5 μ m filter paper, and a 30-ml aliquot sample was frozen from each extraction for later analysis of nitrate and ammonium as described above.

Soil samples for KCl-extractions were mostly from the mineral soil but also included varying amounts of the forest floor, as organic matter fractions determined by loss on ignition ranged from 0.03 to 0.93. Of 72 samples collected, 80% had organic fraction less than 0.4, which would classify them as mineral soil, assuming this corresponds to 20% organic carbon (USDA, 2006).

Data loggers (Onset Computer Corporation) were installed to monitor soil temperature and moisture in three replicates of treatments 1×, 2×, and uncut reference. Treatment 0× plots were not instrumented. Both temperature and soil moisture probes were buried 5–10 cm below the soil surface. The soil moisture sensor measured the dielectric constant of the soil from which volumetric water content was determined. The temperature sensor was a stainless steel 12-bit smart sensor. Temperature and moisture were recorded every 20 min, and average daily values were calculated for a total of 15 months following forest harvesting.

To monitor decomposition rates under field conditions, 15 litterbags of approximately 13 cm × 13 cm were constructed from nylon mesh (3 mm openings) and filled with approximately 54 g (dry weight) of the wood-chip mix. This was equivalent to an application rate of 32 Mg ha⁻¹ or 8 Mg ha⁻¹ more than applied in the 2× treatment. The initial oven-dry weights were estimated based on the moisture content measured in 24 sub-samples of the mix. In the field, all litterbags were placed within a 4 m² area on the forest floor at one of the cut patches (0×) and incubated for one year following the cut. The content of the litterbags was then collected, oven-dried, weighed, ground, and analyzed for C and N using a Carlo Erba NC2500 elemental analyzer.

2.5. Nitrogen retention

To estimate the potential of the wood chips to reduce the inorganic N pulse detected in our patch cut within the first year following cutting, we calculated the excess (i.e., relative to the uncut reference plots) resin-bag or KCl-extracted nitrate or ammonium retained in wood-chip (average of 1× and 2× treatments) plots ($[N]_{\text{chips}} - [N]_{\text{uncut}}$) as a proportion of the excess nitrate or ammonium measured in the untreated (0×) plots ($[N]_{0\times} - [N]_{\text{uncut}}$) following disturbance. This calculation was based on the following equation:

$$\frac{[N]_{0\times} - [N]_{\text{chips}}}{[N]_{0\times} - [N]_{\text{uncut}}} \quad (1)$$

where $[N]_{\text{chips}}$ is the average inorganic N (nitrate or ammonium) concentration measured in 1× and 2× treatments,

$[N]_{\text{uncut}}$ is the average inorganic N concentration measured in uncut reference plots, and $[N]_{0\times}$ is the average inorganic N concentration measured in $0\times$ treatments.

We calculated the inorganic N retention rate for all sampling periods, except for those in which the average inorganic N concentration in $0\times$ treatments was less than in uncut plots (giving a negative denominator to Eq. (1)), which happened in May for nitrate and ammonium in resin bags and in April, May, and July for ammonium in KCl extractions. We report the average inorganic N retention rate both as the average of the rates for all sampling periods and as the rate calculated for the average concentrations.

2.6. Statistical analysis

Measurements of soil parameters taken at each of the six patches prior to forest harvesting were used to assess spatial variability among blocks (cut patches). Based on a one-way ANOVA, pretreatment data indicated that soils at our site did not significantly differ ($p > 0.05$) in pH, cation exchange capacity, exchangeable acidity, or KCl-extractable nitrate and ammonium. These results suggest that soil chemistry in these six patches was not systematically different before treatment, supporting our choice of a randomized complete block model.

Mean nitrate and ammonium concentrations from resin bags located at $0\times$, $1\times$, and $2\times$ wood-chip treated plots were analyzed by analysis of variance (ANOVA) with sub-sampling of experimental units for each sample collection period. The model was $Y_{ijk} = \mu + \tau_i + e_{ij} + d_{ijk}$, where Y_{ijk} = observed response of experimental unit j of treatment i of sub-sample k , μ = grand mean response for all samples, τ_i = fixed effect of treatment i , e_{ij} = random experimental error for the j experimental unit under treatment i , and d_{ijk} = random sampling error for sub sample k from experimental unit j of treatment i . For the KCl-extracts, there was one sample per treatment plot, thus mean nitrate and ammonium concentrations obtained from $0\times$, $1\times$, and $2\times$ wood-chip treated plots were analyzed by a randomized complete block design ANOVA for each sampling interval ($Y_{ij} = \mu + \rho_j + \tau_i + e_{ij}$), where ρ_j = block effect on experimental unit j .

Wood-chip treated plots were compared to uncut plots by two-sample t -tests for both resin bags and KCl-extractions. T -tests were necessary in this design because the uncut plots were not located in the same patch cut designated as a block in the randomized complete block design (Fig. 1).

Comparisons among treatments for both resin bags and KCl extractions were performed by the following linear contrasts. The average of the wood-chip treatments ($1\times$ and $2\times$) were compared to the unamended treatments ($0\times$) to determine whether wood chips succeeded at reducing the availability of nitrate and ammonium in soils. The uncut reference was compared to the $0\times$ treatment to determine whether forest harvesting caused nitrate and ammonium concentrations to rise above those of reference plots. Finally, $1\times$ vs. $2\times$ treatments were compared to determine whether the higher application rate reduced nitrate and ammonium availability in treated plots.

Mean temperature and soil moisture data obtained from treatments $1\times$, $2\times$, and uncut reference plots were analyzed by a randomized complete block design ANOVA ($Y_{ijk} = \mu + \tau_i + \rho_j + e_{ij} + d_{ijk}$) with subsampling of experimental units. Pearson correlations were conducted to determine whether a relationship existed between nitrate and ammonium concentrations in both resin bags and KCl-extractions with changes in soil temperature and moisture.

When significant differences were not detected between treatments, power analysis was conducted at 80% probability of rejecting a false null hypothesis using GPower (Faul and Erdfelder, 1992) to estimate statistically detectable differences among treatments at $\alpha = 0.05$, assuming that the measured means and variances were accurate.

We used regression to assess the potential relationship between organic fraction and KCl extractable nitrate. This analysis was not conducted from resin bags because soil samples were not collected when resin bags were deployed.

For all statistical analyses, differences between treatments were considered significant at $p < 0.05$. However, we acknowledged differences at $p < 0.10$.

3. Results

3.1. Comparison of partial cuts to uncut references

Before reporting the effectiveness of wood chips at mitigating the effects of cutting, we first compare the cut patches without wood chips ($0\times$) to the uncut references, for both resin bags and KCl extractions, to determine when cutting had a significant effect on inorganic N concentrations.

In resin bags, significant differences between $0\times$ and uncut plots were not detected beyond the fall immediately following the cut (Figs. 2 and 3). Nitrate concentrations in $0\times$ plots were 9 times greater than uncut plots for October 2004

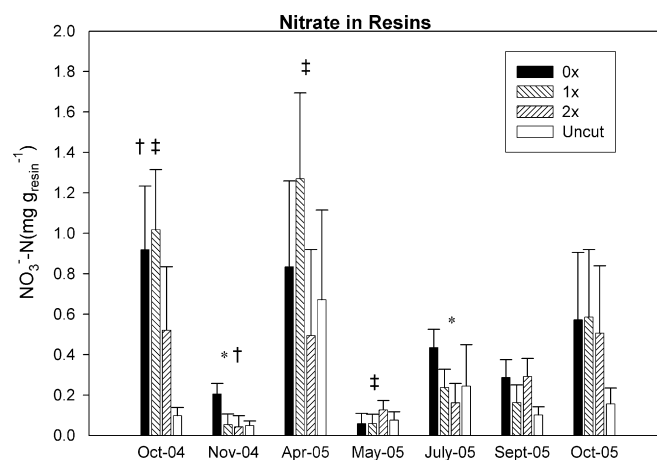


Fig. 2. Mean ammonium concentrations for ion exchange resin bags incubated under field conditions for 7-week intervals. Significant differences ($\alpha = 0.05$) for linear contrasts are denoted by: (*) wood chips (average of $1\times$ and $2\times$) vs. $0\times$. (†) $0\times$ vs. uncut reference. (‡) $2\times$ vs. $1\times$. (+) wood chips (average of $1\times$ and $2\times$) vs. $0\times$ ($\alpha = 0.10$). (§) $0\times$ vs. uncut reference ($\alpha = 0.10$). Absence of symbols indicates no significant difference. Error bars represent standard errors.

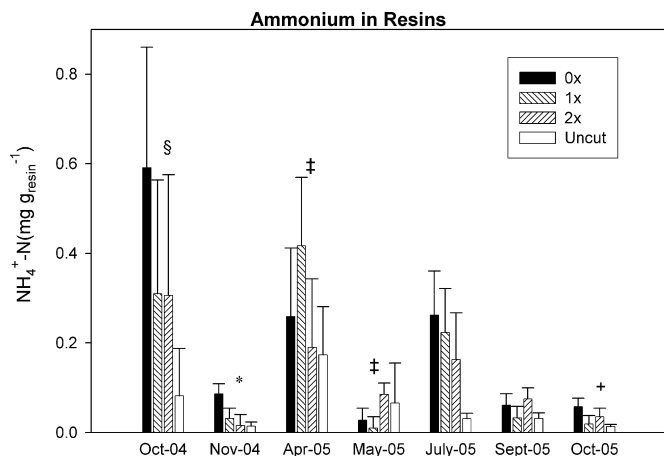


Fig. 3. Mean ammonium concentrations for KCl extracted soil samples collected at 7-week intervals. Significant differences ($\alpha = 0.05$) for linear contrasts are denoted by: (*) wood chips (average of 1x and 2x) vs. 0x. (†) 0x vs. uncut reference. (‡) 2x vs. 1x. (+) wood chips (average of 1x and 2x) vs. 0x ($\alpha = 0.10$). (§) 0x vs. uncut reference ($\alpha = 0.10$). Absence of symbols indicates no significant difference. Error bars represent standard errors.

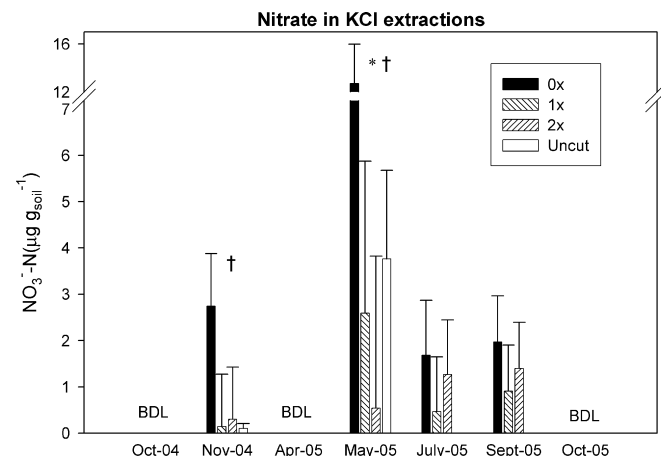


Fig. 4. Mean nitrate concentrations for ion exchange resin bags incubated under field conditions for 7-week intervals. Significant differences ($\alpha = 0.05$) for linear contrasts are denoted by: (*) wood chips (average of 1x and 2x) vs. 0x. (†) 0x vs. uncut reference. (‡) 2x vs. 1x. Absence of symbols indicates no significant difference. Error bars represent standard errors.

(Fig. 2; $p = 0.004$) and 4 times greater during November 2004 (Fig. 2; $p = 0.007$). A year later, in October 2005, the mean nitrate concentration in 0x treatments was 3.7 times the mean in uncut plots. Although differences were large, the high variation in our measurements limited our ability to detect significant differences. A significant effect would have been detected if concentrations in 0x plots had been at least 4.7 times the uncut control. Nitrate concentrations were highest during April 2005 (Fig. 2; $1.3 \text{ mg g}_{\text{resin}}^{-1}$), whereas ammonium peaked earlier, immediately following the cut during October 2004 (Fig. 3; $0.6 \text{ mg g}_{\text{resin}}^{-1}$), when concentrations in 0x treatments were 7 times those in uncut reference plots ($p = 0.05$). Although the mean ammonium concentration in 0x treatments was 8.3 times the mean in the uncut plots in July, we were unable to detect significant differences (Fig. 3). A significant effect would have been detected if concentrations in 0x plots had been at least 12.4 times the uncut control.

Consistent with results from resin bags, KCl extractions indicated that patch cutting had a brief effect on nitrate and ammonium concentrations (Figs. 4 and 5). Nitrate concentrations peaked at about $13 \text{ mg g}_{\text{soil}}^{-1}$ during May 2005 (Fig. 4; treatment 0x) and were significantly greater than in uncut plots in both November 2004 and May 2005, while concentrations during spring snowmelt (April 2005) were below $0.15 \text{ mg g}_{\text{soil}}^{-1}$, our detection limit (Fig. 4). Nitrate concentrations in 0x treatments were 27 times the concentrations detected for uncut plots during November 2004 (Fig. 4; $p = 0.048$) and 3 times those detected in uncut plots for May 2005 (Fig. 4; $p = 0.046$). Ammonium concentrations in the 0x treatment peaked at about $24 \text{ mg g}_{\text{soil}}^{-1}$ during October 2004 (Fig. 5), and were double the concentrations detected in uncut plots ($p = 0.05$). At the other sampling dates, the uncut reference was just as likely to be higher in ammonium than the 0x treatments; none of the differences were statistically significant (Fig. 5).

3.2. Wood-chip effects on soil nitrate concentrations

The wood-chip application lowered nitrate concentrations measured with resin bags by an average of 30% (Eq. (1), using average concentrations) or 42% (Eq. (1), using average percentage across sampling periods), compared to 0x treatments. For KCl extracts, the nitrate retention rate was 100% (Eq. (1), using average concentrations) or 78% (Eq. (1), using average percentage across sampling periods). For the fall after forest harvest (October and November 2004) nitrate concentrations detected in resin bags (Fig. 2) and KCl-extracted soils (Fig. 4) varied through time. In October 2004, 7 weeks after forest harvest, nitrate concentrations in KCl-extracted soils for all treatments were below the instrument detection

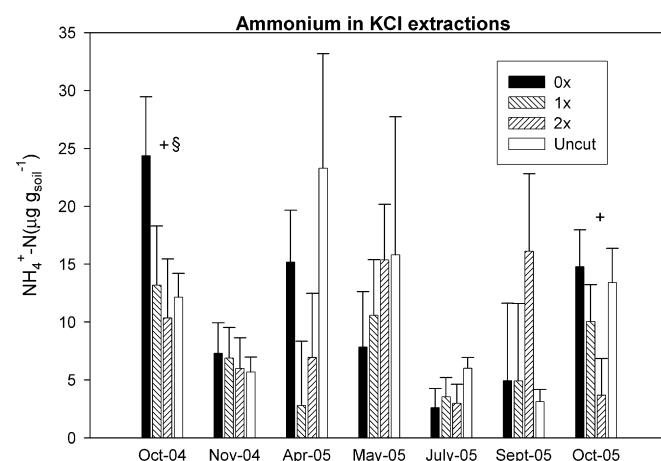


Fig. 5. Mean nitrate concentrations for KCl extracted soil samples collected at 7-week intervals. Significant differences ($\alpha = 0.05$) for linear contrasts are denoted by: (*) wood chips (average of 1x and 2x) vs. 0x. (†) 0x vs. uncut reference. (‡) 2x vs. 1x. (+) wood chips (average of 1x and 2x) vs. 0x ($\alpha = 0.10$). (§) 0x vs. uncut reference ($\alpha = 0.10$). Absence of symbols indicates no significant difference. Error bars represent standard errors. BDL = below detection limit.

limit (Fig. 4; $0.15 \text{ mg g}_{\text{soil}}^{-1}$). For resin bags in October 2004, wood chips retained 18% of the increased nitrate in $0\times$ treatments compared to uncut stands (Fig. 2; Eq. (1)). However, the effect of wood chips ($1\times$ and $2\times$) in resin bags was not statistically significant in October 2004 (Fig. 2; $p = 0.44$); the mean of the $1\times$ treatment was greater than that of the $0\times$. During November, after leaf fall, nitrate concentrations measured with KCl-extractions in wood-chip treated plots averaged only 8% of the concentrations in the $0\times$ treatment soils (Fig. 4; $p = 0.12$) and 25% of concentrations measured using resin bags (Fig. 2; $p = 0.0001$). At this date, for resin bags, 101% of the increased nitrate in $0\times$ treatments compared to uncut stands was retained by the woodchips and 95% was retained for KCl extractions (Eq. (1); Figs. 2 and 4).

During spring snowmelt (9 April 2005), nitrate concentrations were high for all treatments in resin bags and below detection limit for KCl extractions (Figs. 2 and 4). For resin bags, the difference between the woodchip treatments and the $0\times$ treatment was not significant ($p = 0.82$), and again, the mean of the $1\times$ treatment was greater than that of the $0\times$. The retention rate (Eq. (1)) was negative for resin bags, but not statistically significant, with wood-chip treated plots having 29% higher nitrate concentrations than the untreated cut plots minus the controls (Fig. 2).

During the growing season following harvest (30 May 2005, 17 July 2005, 5 September 2005), nitrate concentrations in May for KCl-extracts of soil from wood-chip treated plots averaged only 13% of those measured in the $0\times$ treatment (Fig. 4; $p = 0.03$) and 46% of the $0\times$ treatment for resin bags during July (Fig. 2; $p = 0.01$). For KCl extractions, the nitrate retention rate was 125% for May (Fig. 4; Eq. (1)) but not for resin bags as the mean nitrate concentration in uncut references was higher than that in $0\times$ treatments (Fig. 2). In July, however, wood chips retained 124% (Eq. (1)) of the increased nitrate in $0\times$ treatments for resin bags compared to uncut stands (Fig. 2) and 49% of the increased nitrate in $0\times$ treatments for KCl extractions (Fig. 4). For KCl extractions in uncut plots during July, concentrations were below the instrument detection limit (Fig. 4). In September, wood-chip treated plots in resin bags and KCl extractions retained 32% and 42% (Eq. (1)), respectively, of the increased nitrate in $0\times$ treatments compared to uncut stands, but there were no significant differences detected among treatments (Figs. 2 and 4) and concentrations in uncut plots for KCl extractions were again below detection limit (Fig. 4). Likewise, for the last sample collection during the fall one year after harvesting (23 October 2005), no significant differences were detected among treatments for resin bags (Fig. 2) and wood chips had retained only 6% (Eq. (1)) of the increased nitrate for resin bags in $0\times$ treatments compared to uncut stands (Fig. 2). Nitrate concentrations for October 2005 in KCl extractions were below detection limit for all treatments (Fig. 4). Given the high variance in resin bags, an effect of woodchips on inorganic N concentrations in October 2005 would not be detectable unless woodchips reduced nitrate concentrations to 5% of that in $0\times$ treatments (Fig. 2). In other words, we can be confident only that nitrate concentrations were not reduced by more than 95% by the wood-chip treatment.

The $2\times$ treatment application was not consistently better at lowering nitrate concentrations compared to the $1\times$ wood-chip application (Figs. 2 and 4). Because of high variability in the soil KCl-extractions (Fig. 4), no significant differences could be detected between the $1\times$ and $2\times$ treatments, even if $2\times$ concentrations were to fall below the detection limit. In contrast, data from resin bags indicate that during October 2004 (Fig. 2; $p = 0.03$) and April 2005 (Fig. 2; $p = 0.004$), nitrate concentrations in $2\times$ treatments were significantly less than those measured in $1\times$ treatments. However, during May 2005, nitrate concentrations in resin bags from the $2\times$ treatment were significantly greater than those in the $1\times$ treatment (Fig. 2; $p = 0.02$). Nitrate concentrations detected for resin bags in May were relatively low for all treatments compared to other sampled months (Fig. 2), such that the higher nitrate concentrations detected in $2\times$ treatments were not of sufficient magnitude to be of concern.

3.3. Wood-chip effects on soil ammonium concentrations

The wood-chip application lowered ammonium concentrations measured by resin bags by 42% (Eq. (1), using average concentrations) or 36% (Eq. (1), using average percentage across sampling periods). For KCl extracts, the ammonium retention rate was 93% (Eq. (1), using average concentrations) or 107% (Eq. (1), using average percentage across sampling periods). Ammonium concentrations detected in resin bags and KCl-extracted soils varied through time and the effect of wood chips was sometimes significant (Figs. 3 and 5).

During the fall immediately following harvest, ammonium concentrations in wood-chip treated plots were about half those of the $0\times$ treatment in October (Figs. 3 and 5; $p = 0.10$ for resin bags and 0.09 for KCl-extractions) with a retention rate of 56% for resin bags and 103% for KCl extractions (Eq. (1)). In November, wood-chip treated plots averaged 28% of the ammonium concentrations measured by resin bags in the $0\times$ treatment (Fig. 3; $p = 0.02$); the ammonium retention rate was 87% for resin bags (Eq. (1)). The effect of wood chips in November measured by KCl extractions was 53% but not statistically significant (Fig. 5).

During spring (9 April 2005), ammonium concentrations in wood-chip treated plots were not significantly different from those in the $0\times$ treatment for either resin bags or KCl extracts (Figs. 3 and 5). The calculated retention rate, in resin bags, was negative (52%; Eq. (1)). The $1\times$ treatment had much lower ammonium in KCl extractions than the $2\times$, but this difference was not significant (Fig. 5; $p = 0.16$). However, for resin bags, ammonium concentrations in the $2\times$ treatment were significantly less than those in the $1\times$ treatment (Fig. 3; $p = 0.02$). Eq. (1) was not applicable for KCl extractions as uncut treatments had higher ammonium concentrations than $0\times$ treatments, but this difference was not significant (Fig. 5).

For the growing season following harvest, ammonium concentrations were not significantly different between wood-chip treated plots and $0\times$ treatments for resin bags or for KCl extractions (30 May 2005, 17 July 2005, and 5 September 2005; Figs. 3 and 5). As with nitrate, Eq. (1) was not applicable in

May for either resin bags or KCl extractions, because the mean in uncut references was greater than that in 0× plots (Figs. 3 and 5). For the remainder of the growing season, ammonium in resin bags in wood-chip plots was reduced by 30% in July and 24% in September (Eq. (1)), but these differences were not significant (Fig. 3). For KCl extractions, Eq. (1) was also not applicable in July and was negative for September 2005 (309%; Fig. 5), but again the differences were not significant (Fig. 5). In October 2005, ammonium concentrations in wood-chip treated plots were about half those of the 0× treatment for both resin bags (Fig. 3; $p = 0.05$) and KCl-extracted soils (Fig. 5; $p = 0.09$), giving an ammonium retention rate of 69% for resin bags and 582% for KCl extractions (Figs. 3 and 5; Eq. (1)).

Like nitrate, ammonium concentrations were not consistently less in the 2× treatment than in the 1× treatment (Figs. 3 and 5). In fact, in May 2005, the 2× wood-chip application had higher ammonium concentrations than the 1×, significantly so for resin bags (Fig. 3; $p = 0.01$). For KCl extractions, detectable differences varied by sampling date; significant differences would have been detectable with an average ammonium reduction of 16% in 2× treatments from concentrations detected in 1×, except for April and September, in which reductions to below detection limit would have yielded insignificant results (Fig. 5).

3.4. Soil moisture, temperature, and organic matter content

Soils in 1× plots were drier than in 2× and uncut plots (Fig. 6; $p < 0.0001$). For uncut plots, the average volumetric moisture content was 0.084, while for wood-chip treated plots the value was 0.072 in the 1× treatment and 0.084 in the 2× treatment. Evidently, the additional wood chips applied to the 2× treatment reduced evaporative losses compared to the 1× treatment, and this was sufficient to compensate for the loss of canopy, as soil moisture was indistinguishable between 2× and uncut plots (Fig. 6; $p = 0.97$).

Soil temperature at a depth of approximately 5 cm below the soil surface averaged 8.6 °C in uncut reference plots, 9.1 °C at 1× treatment plots, and 9.0 °C at 2× treatment plots (Fig. 7).

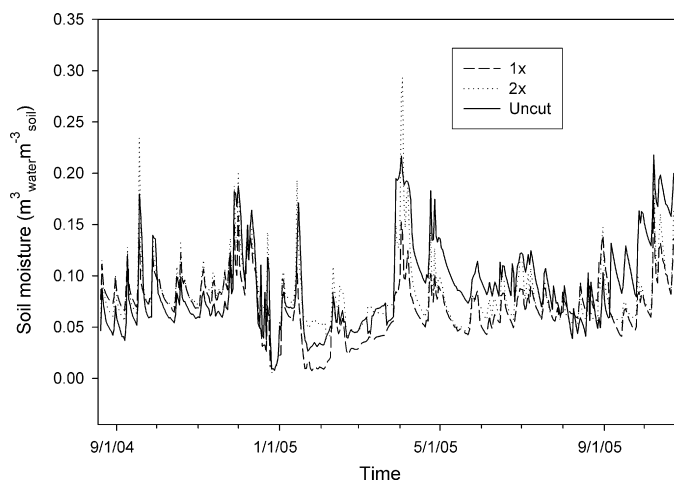


Fig. 6. Mean daily soil moisture for wood-chip treated plots and uncut sites. Data were recorded from August 20, 2004 through October 23, 2005.

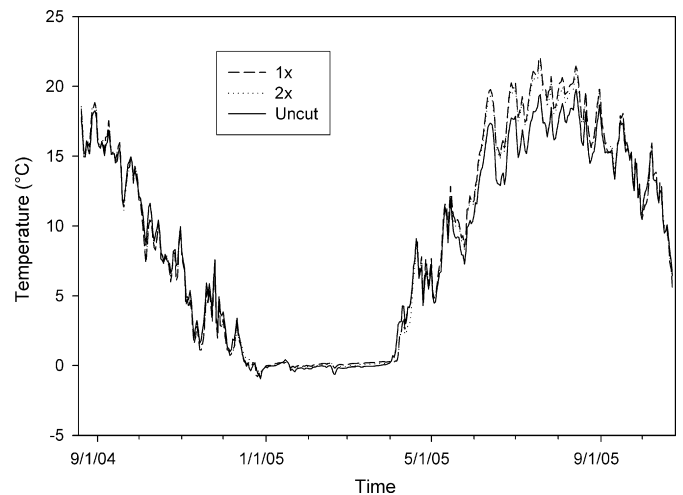


Fig. 7. Mean daily soil temperature for wood-chip treated and uncut plots. Data were recorded from August 20, 2004 through October 23, 2005.

Results from the ANOVA indicate that there were no significant soil temperature differences among treatments for the duration of the experiment ($p = 0.57$). However, paired *t*-tests conducted for the growing season only (May 1 through October 23, 2005), indicate that soils in both the 1× and 2× treatments had significantly higher temperatures than those in uncut plots (Fig. 7; $p = 0.002$ and $p = 0.009$ respectively). Soil temperatures between 1× and 2× plots were statistically indistinguishable ($p = 0.64$) despite temperatures in 1× plots being among the highest recorded during summer (Fig. 7). Harvested plots without wood chips (0×) were not instrumented. We used Pearson correlations to test whether temperature and moisture explained some of the variation in N dynamics detected at our plots. However, correlations between soil temperature and moisture and nitrate and ammonium concentrations in both resin bags and soil extractions were weak and not statistically significant. The strongest relationship was a negative correlation of ammonium concentrations in resin bags and soil moisture ($r = -0.32$; $p = 0.15$).

Soil samples for KCl extractions reflected a variable mix of organic and mineral horizons, and we expected that this variation would affect our measurements of nitrate and ammonium. The relationship between the organic content of the samples and the ammonium concentrations was significant but weak ($R^2 = 0.08$; $p = 0.01$), and the relationship between nitrate concentrations and soil organic content was weak and not significant ($R^2 = 0.002$; $p = 0.7$).

3.5. Wood-chip incubations and litterbags

In general, the wood-chip mix released little nitrate when incubated in the laboratory. Cumulative net nitrification rates (\pm S.D.) for the incubated wood chips were 0.13 (± 0.16) mg N kg⁻¹ chips after 7 days, 0.40 (± 0.25) mg N kg⁻¹ chips after 17 days, and 0.22 (± 0.19) mg N kg⁻¹ chips after 29 days. In other words, chips began immobilizing nitrate between 17 and 29 days, in these conditions.

In the field, wood chips lost an average of 19% (S.D. ± 6) of their initial dry weight in nylon mesh bags incubated for one year. On average, the initial wood-chip mix was 0.4% (± 0.04) N, and 50 (± 0.7)% C. After field incubation, the wood-chip mix almost doubled in N concentration to about 0.7% (± 0.1) of the dry weight of chips while C remained relatively constant at 49 (± 2)%. The initial C:N ratio of the wood-chip mix was 125:1 and the final was 70:1 (± 16) after one year of decomposition in the field.

Based on the decomposition rates of wood chips in litterbags, we estimated the potential of the wood-chip application to immobilize N for the first year following application. We subtracted the initial concentration of N in wood chips (0.4%) from the N concentration in wood chips after field incubation (0.7%) and corrected for a 20% wood-chip mass loss. Our estimates suggest that the 1 \times wood-chip application (12 Mg ha⁻¹) could have immobilized 19 kg N ha⁻¹ while the 2 \times (24 Mg ha⁻¹) could have immobilized 38 kg N ha⁻¹.

4. Discussion

4.1. Mechanisms for N gain in wood chips

There are many possible sources for the nitrogen gained by the surface-applied wood chips, such as throughfall (Downs et al., 1996), insect frass (Bocock, 1963), N fixation (Granhall and Lindberg, 1977), passive diffusion from the organic soil layer (Berg, 1988), and fungal translocation from the mineral soil and forest floor (Hart and Firestone, 1991; Setälä et al., 1996; Zeller et al., 2000). Nitrogen deposition in the Catskill Mountains (6–7 kg N ha⁻¹ year⁻¹) can explain only about 11–13% of the N gained by the chips. Other studies of surface residue decomposition have found fungal hyphal translocation of N to explain most or all of the N assimilated (Hart and Firestone, 1991; Frey et al., 2000).

Because fungi grow efficiently in nutrient poor and recalcitrant substrates (Paustian and Schnürer, 1987) and can translocate nutrients upward from the mineral soil (Frey et al., 2000, 2003), they can take advantage of the N-rich soil that is typical after forest disturbance, which further promotes fungal translocation of nutrients (Frey et al., 2003). Thus, we propose that because the wood chips were in direct contact with soils of elevated extractable nitrate and ammonium concentrations after cutting, fungal hyphae and/or microbial communities translocated nutrients into (Berg and Söderström, 1979; Hart and Firestone, 1991; Zeller et al., 2000) and within (Schimel and Hättenschwiler, 2007) the chips, resulting in N immobilization.

4.2. Nitrogen immobilization potential

From N measurements in resin bags we estimated an average reduction in nitrate and ammonium in wood-chip treated plots of about a third of the elevation in concentrations measured in unamended plots compared to uncut reference treatments. However, for KCl extractions, we estimated average nitrate reductions of 78% or 100%. Whether this reduction in nitrate

and ammonium in resin bags would translate into an equivalent reduction in stream export after harvesting could not be determined in this study. Previous studies in the Catskills suggest that water percolating through the O-horizon during large storm events contributes significantly to stream flow (Brown et al., 1999), and given a high correlation between soil solution and stream chemistry (Burns and Murdoch, 2005), we expect that nitrate export would be reduced as a result of wood-chip application. Our estimates are supported by the low nitrification rates in laboratory-incubated wood chips and significant N immobilization in field-incubated litterbags, which suggest that over a 1-year period, the chips could potentially leach 0.068 kg N ha⁻¹ but immobilize 38 kg N ha⁻¹ (based on a 2 \times treatment; C:N reduced from 125:1 to 70:1). This rate is similar to the reported immobilization rate in red-maple wood-chip litterbags (19–29 kg N ha⁻¹; 15–20 g of material in 25 cm \times 30 cm bags) under ambient conditions at the Harvard Forest in Massachusetts, USA (Micks et al., 2004).

If net N mineralization begins at a C:N ratio of about 30:1 (Berg and Staaf, 1981), then the C content of the wood chips was sufficient to continue to immobilize N well past the first year following application. Assuming a continued wood-chip decomposition rate similar to that reported by Micks et al. (2004), we would expect to find a C:N ratio of approximately 35:1 in our 2 \times treatment after two years (30% wood-chip mass loss and 150% N gain), representing an additional 106 kg of N sequestered per hectare. For comparison, nitrate export from a 24-ha clearcut approximately 2-km east of our site was 37 kg N ha⁻¹ in the first year (Burns and Murdoch, 2005).

Caution is warranted in scaling N immobilization rates from litterbags to plots or stands. Litterbags do not allow wood chips as much contact with soil compared with field applications, which might be thought to limit translocation of N into the chips. On the other hand, at some application rate, the availability of N must limit immobilization rates. Although our litterbag measurements suggest that sufficient C was available to immobilize the first-year nitrate peak at the adjacent clearcut, results from resin bags suggested an average nitrate retention rate of only 30% or 42% (Eq. (1)). This disagreement is not surprising, because according to studies that have traced changes in the isotopic composition of nitrate in precipitation and stream water in the Catskills (Burns and Kendall, 2002) and in Vermont (Ohte et al., 2004), nitrate can bypass biological processes in the soil and be transported unaltered through soils. Since the wood chips did not completely cover the soil surface (especially in the 1 \times treatment), it seems reasonable that some nitrate could reach our resin bags without having interacted with the wood chips, resulting in low estimates of inorganic N reduction. The average nitrate retention rate calculated for KCl extractions was higher, at 78% or 100% (Eq. (1)), but this measure may be less indicative of the potential delivery of nitrate to streams (see Section 4.4, below).

Addition of organic matter to the forest floor can also induce denitrification. In forest soils of the Catskill region, denitrification appears to be limited by water-filled pore space, carbon, and nitrate supply (Ashby et al., 1998). The application of wood

chips to the forest floor retains soil moisture and releases dissolved organic carbon. Because forest harvesting generally increases nitrate supply, this may create conditions suitable for denitrification. Studies of Catskill soils have reported that denitrification can be greater in soils amended with glucose and nitrate than with nitrate alone (2 and 1 $\mu\text{g N}_2\text{O-N kg}^{-1} \text{h}^{-1}$ respectively; Ashby et al., 1998). Similarly, at a farm in New Zealand, the presence of labile carbon in a denitrification wall of soil mixed with saw dust promoted anaerobic conditions conducive to denitrification (3.6 $\text{g N m}^{-3} \text{d}^{-1}$) of contaminated groundwater (Schipper and Vojvodic-Vukovic, 1998). Thus, it is possible that addition of material high in C:N to the forest floor after harvesting reduces nitrate concentrations by promoting denitrification as well as N immobilization.

4.3. Implications for forest management

High spatial variability in nitrate and ammonium concentrations among our measurements limited our ability to detect treatment effects. Some of this variability may have been due to the small size of our wood-chip treated plots (9.2 m \times 9.2 m) and cut patches (approximately 0.1 ha). The roots and canopies of adjacent uncut trees extended to varying degrees into our experimental plots, which undoubtedly influenced the N dynamics within the plots. Another source of variation is the rockiness of the soil, which likely caused large variations in the water flux and thus the inorganic N adsorbed by the resin bags. The variable depth of organic horizons meant that resin bags and KCl-extractions buried and collected at 5–10 cm depth represented varying contributions from organic and/or mineral horizons. However, we found no relationship between nitrate and ammonium concentrations and organic matter content in the soils sampled, which suggest that this source of variation did not strongly affect the measurements.

Because of the high variability in our nitrate and ammonium measurements, it remains unclear how much more inorganic N could be immobilized by the addition of more chips (2 \times treatment). A 2 \times application would be impractical over large areas because the doubled rate requires wood chips to be introduced from other sources; our 1 \times treatment used only the chips generated by application of all slash smaller than 20 cm in diameter, in this case, 12 Mg ha^{-1} . Higher rates could be applied in limited areas of the landscape such as in sensitive riparian zones. Further research is needed before an application rate can be recommended.

We tested the potential of this practice using patch cuts in only one stand in the Catskill region of New York, an area of elevated nitrogen deposition with demonstrated water quality changes associated with forest harvesting (Ollinger et al., 1993; Lovett and Rueth, 1999); testing of this practice in different areas or after different silvicultural treatments is warranted. Additions of material with a high C:N ratio have been reported to promote N immobilization in many other ecosystems (Mazzarino et al., 1993; Takahashi and Ikeda, 1994; Hobbie and Chapin, 1996; Gallardo and Merino, 1998). However, more information would be helpful on the effects of the C:N ratio of the chipped material, chip decomposition rates, and the extent

and duration of the improvement in stream-water quality. Wood-chip application should be effective in other silvicultural treatments. Because the magnitude of nitrate leaching losses are dependent on the intensity of the harvest (Wang et al., 2006), the rate of wood-chip application would need to be adjusted based on the intensity of the cut.

A comparison of inorganic N concentrations in samples of the upper 5–10 cm of soil at our patch cut to those of soils of the O-horizon at an adjacent 24-ha clearcut studied by Burns and Murdoch (2005) demonstrates the capacity of residual living trees to reduce nutrient losses following disturbance. Mean KCl-extractable nitrate concentrations from April through November in O-horizon soils in the clearcut (about 35 mg kg^{-1}) for the first year after the cut were approximately 16 times greater than the mean nitrate concentrations in 0 \times treatments for the first year of our experiment (2.2 mg kg^{-1} ; Fig. 4). For KCl-extractable ammonium, mean values from April through November (about 60 mg kg^{-1}) were about 4 times greater than in our 0 \times treatments (14.8 mg kg^{-1} ; Fig. 5). Similarly, average nitrate (3.0 $\text{mg N g}_{\text{resin}}^{-1}$) and ammonium (0.25 $\text{mg N g}_{\text{resin}}^{-1}$) concentrations obtained from resin bags for the first year after harvesting in the 24-ha clearcut were greater than those at our cut patches (0.47 $\text{mg N g}_{\text{resin}}^{-1}$ for nitrate and 0.19 $\text{mg N g}_{\text{resin}}^{-1}$ for ammonium; Figs. 2 and 3). The lower nitrate and ammonium concentrations detected in cut patches are consistent with previous work that indicates that partial cuts typically result in lower export of nitrate and ammonium to streams than clearcuts (Messina et al., 1997; Wheeler et al., 2000; Wang et al., 2006). Thus, we hypothesize that in areas sensitive to forest harvesting, a practice that combines partial cutting with a wood-chip treatment should maintain higher water quality than applying either practice alone.

Our study did not evaluate the cost of implementing such a BMP, which depends not only on the chipping process (i.e., fuel use for operation and transportation, operator costs, etc.) but also on the competing demands for slash. The current interest in woody biomass as an alternative source of energy could increase this opportunity cost. At the time of our study, there were no viable markets for the top wood we chipped.

4.4. Ion-exchange resin bags vs. soil KCl-extractions

Soil KCl-extractions and resin bags provided conflicting answers regarding the availability of nitrate and ammonium in experimental plots. For example, during the months of October 2004 and April and October 2005, nitrate concentrations in KCl extractions were below detection limit for all treatments (Fig. 4), while resin bags gave the highest concentrations for all treatments for the length of our study (Fig. 2).

Ion-exchange resin bags collect nitrate and ammonium that is moving downward, and possibly laterally, through the soil during rain events over the period of burial (in our case, 7 weeks). In other words, the resin bags provide information on the mobility of nitrate and ammonium in the soil. They may also adsorb soil nitrate and ammonium that are in contact with the resin given sufficient moisture. In contrast, soil KCl-extractions provide an operationally defined measurement of exchangeable nitrate and ammonium at the time of soil collection.

In general, our soil KCl extractions indicated that ammonium concentrations at our plots were greater than nitrate, while for resin bags, nitrate concentrations were greater than ammonium. A greater concentration of ammonium than nitrate detected in soil KCl-extractions likely reflects the true concentration in soils where nitrified nitrate is readily transported out of the soil to streams during rain events. Conversely, greater nitrate than ammonium concentrations are expected in resin bags because they reflect this greater flux in soil solution.

4.5. Possible complications and stand regeneration

Chipped slash contains N. The amount of N added to our plots in wood chips was 48 kg ha⁻¹ for the 1× treatment and 96 kg ha⁻¹ for the 2×, based on the initial C:N ratios of the chip mix (125:1). To understand how much N could be leached to the forest floor from chips, we measured the net nitrification potential of our chip mix in the lab (0.008 mg N kg_{chips}⁻¹ day⁻¹; based on a 29 day incubation). These rates were only 0.8% of the rates typically observed for one-month incubations of O-horizon soils in the Catskill region (about 1 mg N kg_{soil}⁻¹ day⁻¹; Burns and Murdoch, 2005), which suggests that the amount of nitrate potentially released by the wood-chip mix is much less than that released by local soils. These initial net nitrification rates (29 days) probably reflect the decomposition of material low in C:N included in the wood-chip mix, such as the leaves of cut trees that were ground during wood-chipping.

The application of wood chips may affect forest regeneration. Mulches are widely used in urban forestry as a way to limit germination and suppress the growth of weeds (Skroch et al., 1992). In our study, both the 1× and 2× wood-chip applications were rather thin (5–10 mm) and did not completely cover the forest floor, which was visible through the wood chips. It seems unlikely that the application of wood chips at rates tested in our study would interfere with the regeneration process. In fact, a thin layer of wood chips may be beneficial for regeneration; in our study, more moisture was retained in the 2× treatment than the 1× treatment ($p < 0.0001$).

Mulches are also known to lower nutrient availability through immobilization (Waldrop et al., 2003), which was the intent of our wood-chip application. During our experiment, nitrate and ammonium concentrations in soils (upper 5–10 cm) under wood-chip treated cut plots were generally less than in untreated cut plots (0×), though not less than in uncut reference plots (Figs. 2–5). Some species, such as pin cherry, germinate in response to elevated soil inorganic N concentrations (Auchmoody, 1979), and delaying the regeneration of early successional species may slow ecosystem recovery and the return of nutrient cycling to steady-state (Likens and Bormann, 1995). However, regeneration at our site was limited by deer herbivory rather than by nutrient limitation, which precluded an assessment of the effects of wood chips on regeneration, because only a few beech stems survived deer browse. Since slash piles are known to protect seedlings from deer browsing (Grisez, 1960), chipping all available slash may not be advantageous to regeneration in areas of dense deer popula-

tions. To determine the effects of wood-chip application on regeneration would require studies in areas of lower deer density or exclusion of deer. The effect of wood-chip application on forest regeneration is a topic worthy of study before this practice can be more widely recommended as a water-quality BMP.

4.6. Questions for future research

Future research on the potential of wood chips as a forestry BMP should measure stream chemistry to assess the success of the practice at protecting water quality. There would also be value in knowing whether wood-chip application on the forest floor promotes denitrification. Future studies could also develop recommendations for landscape application since it is currently unclear whether wood chips would most effectively be spread over entire large cuts, patches, or in riparian areas.

5. Conclusions

The results of this study suggest that the application of wood chips to the forest floor after forest harvesting has the potential to immobilize some of the inorganic N that would otherwise be available for leaching during the first year after harvesting. The wood chips reduced the average nitrate by 30% to 40%, according to resin bags, and by 80% to 100%, based on KCl extracts. We also observed a rapid accumulation of N in the chips, which supports our claim that wood chips have the potential to immobilize a significant amount of N.

The high variability of nitrate and ammonium concentrations measured in soil extracts and resin bags limited our ability to quantify the effects of the wood-chip application. Further study of wood-chip application as a potential new BMP is clearly warranted. We contend that application of wood chips at sufficient rates should diminish the deleterious effects of forest harvesting on surface water quality that have been demonstrated in regions that receive high rates of atmospheric N deposition.

Acknowledgements

We wish to thank David Fermoile and Tom Grocholl from the Frost Valley YMCA for providing the research site and their invaluable assistance. Pete Murdoch, Don Bickelhaupt, Amber Knowlden, Carolyn Griffin, and Sarah Rienhardt helped with laboratory analysis, and Steve Stehman and Eddie Bevilacqua helped with statistical analysis. Jeff Dietrich provided help in the field and Gabriela and Liliana Mendoza assisted during sample collection and experimental setup.

This research was funded in part by an NSF-GK12 fellowship, a Sigma Xi research society grant, and by the USDA Cooperative Research Service.

References

- Aber, J.D., Ollinger, S.V., Driscoll, C.T., Likens, G.E., Holmes, R.T., Freuder, R.J., Goodale, C.L., 2002. Inorganic N losses from a forested ecosystem in

- response to physical, chemical, biotic and climatic perturbations. *Ecosystems* 5, 648–658.
- Ashby, J.A., Bowden, W.B., Murdoch, P.S., 1998. Controls on denitrification in riparian soils in headwater catchments of a hardwood forest in the Catskill Mountains, USA. *Soil Biol. Biochem.* 30, 853–864.
- Auchmoody, L.R., 1979. Nitrogen fertilization stimulates germination of dormant pin cherry seed. *Can. J. Forest Res.* 9, 514–516.
- Baldigo, B.P., Murdoch, P.S., Burns, D.A., 2005. Stream acidification and mortality of brook trout (*Salvelinus fontinalis*) in response to timber harvest in three small watersheds of the Catskill Mountains, New York, USA. *Can. J. Fisher. Aquat. Sci.* 62, 1168–1183.
- Berg, B., 1988. Dynamics of nitrogen (^{15}N) in decomposing Scots pine (*Pinus sylvestris*) needle litter: long-term decomposition in a Scots pine forest. *VI. Can. J. Bot.* 66, 1539–1546.
- Berg, B., Söderström, B., 1979. Fungal biomass and nitrogen in decomposing Scots pine needle litter. *Soil Biol. Biochem.* 11, 339–341.
- Berg, B., Staaf, H., 1981. Leaching, accumulation and release of nitrogen in decomposing forest litter. In: Clark, F.E., Rosswal, T. (Eds.), *Terrestrial Nitrogen Cycles*. Swedish Natural Science Resources Council, Stockholm, pp. 163–178.
- Binkley, D., Matson, P., 1983. Ion exchange resin bag method for assessing forest soil nitrogen availability. *Soil Sci. Soc. Am. J.* 47, 1050–1052.
- Binkley, D., Hart, S.C., 1989. The components of nitrogen availability assessments in forest soils. In: Stewart, E.D. (Ed.), *Advances in Soil Science*, vol. 10. Springer-Verlag, New York, pp. 57–122.
- Bocock, K., 1963. Changes in the amount of nitrogen in decomposing leaf litter in sessile oak (*Quercus petraea*). *J. Ecol.* 51, 555–556.
- Brown, V.A., McDonnell, J.J., Burns, D.A., Kendall, C., 1999. The role of event water, a rapid shallow flow component, and catchment size in summer stormflow. *J. Hydrol.* 217, 171–190.
- Burns, D.A., Kendall, C., 2002. Analysis of delta ^{15}N and delta ^{18}O to differentiate NO_3^- sources in runoff at two watersheds in the Catskill Mountains of New York. *Water Resour. Res.* 38 (5), 1051–1061, doi:10.1029/2001WR000292.
- Burns, D.A., Murdoch, P.S., 2005. Effects of a clearcut on the net rates of nitrification and N mineralization in a northern hardwood forest, Catskill Mountains, New York, USA. *Biogeochemistry* 72, 123–146.
- Christ, M.J., Peterjohn, W.T., Cumming, J.R., Adams, M.B., 2002. Nitrification potentials and landscape, soil and vegetation characteristics in two Central Appalachian watersheds differing in NO_3^- export. *Forest Ecol. Manage.* 159, 145–158.
- Dahlgren, R.A., Driscoll, C.T., 1994. The effects of whole-tree clear-cutting on soil processes at the Hubbard Brook Experimental Forest, New Hampshire, U.S.A. *Plant soil* 158, 239–262.
- Downs, M., Nadelhoffer, K., Melillo, J., Aber, J., 1996. Immobilization of a ^{15}N -labelled nitrate addition by decomposing forest litter. *Oecologia* 105, 141–150.
- Driscoll, C.T., Schaefer, D.A., 1989. Overview of nitrogen processes. In: Malanchuk, J.L., Nilsson, J. (Eds.), *The Role of Nitrogen in the Acidification of Soils and Surface Waters*, vol. 10. Nordic Council of Ministry, Copenhagen, pp. 4.1–4.12.
- Eaton, A.D., Clesceri, L.S., Greenberg, A.E., 1995. *Standard Methods for the Examination of Water and Wastewater*, 19th ed. American Public Health Association, New York.
- EPA, 2002. List of drinking water contaminants (MCLs [Online] <http://www.epa.gov/safewater/mcl.html#mcls> (posted February 28, 2006; verified 3-18-06).
- Faul, F., Erdfelder, E., 1992. GPOWER: A priori, Post-hoc, and Compromise Power Analyses for MS-DOS [Computer program]. Bonn University, Department of Psychology, Bonn, FRG.
- Fenn, M.E., Poth, M.A., Aber, J.D., Baron, J.S., Bormann, B.T., W, J.D., Lenly, A.D., McNulty, S.G., Ryan, D.E., Stottlenmyer, R., 1998. Nitrogen excess in North American ecosystems: predisposing factors, ecosystem responses, and management strategies. *Ecol. Appl.* 8, 706–733.
- Firda, G.D., R. Lumia, R. Murray, P.M., Freeman, W.O. 1995. *Water Resources Data, New York, Water Year 1994*. NY-94-1. U.S. Geological Survey, Albany.
- Frey, S., Six, J., Elliot, E., 2003. Reciprocal transfer of carbon and nitrogen by decomposer fungi at the soil-litter interface. *Soil Biol. Biochem.* 35, 1001–1004.
- Frey, S., Elliot, E., Paustian, K., Peterson, G., 2000. Fungal translocation as a mechanism for soil nitrogen inputs to surface residue decomposition in a no-tillage agroecosystem. *Soil Biol. Biochem.* 32, 689–698.
- Gallardo, A., Merino, J., 1998. Soil nitrogen dynamics in response to carbon increase in a mediterranean shrubland of SW Spain. *Soil Biol. Biochem.* 30, 1349–1358.
- Goodale, C.L., Aber, J.D., McDowell, W.H., 2000. The long-term effects of disturbance on organic and inorganic nitrogen export in the White Mountains, New Hampshire. *Ecosystems* 3, 343–450.
- Granhall, U., Lindberg, T., 1977. Nitrogen fixation at coniferous forest sites within the Swecon project. *Swedish Conif. Forest Project Tech. Rep.* 11, 1–39.
- Grisez, J., 1960. Slash helps protect seedlings from deer browsing. *J. Forestry* 58, 385–387.
- Hart, S.C., Firestone, M.K., 1991. Forest floor-mineral soil interactions in the internal nitrogen cycle of an old-growth forest. *Biogeochemistry* 12, 103–127.
- Hendrickson, O.Q., Chatarpaul, L., Burgess, D., 1989. Nutrient cycling following whole-tree and conventional harvest in northern mixed forest. *Can. J. Forest Resour.* 19, 725–735.
- Hobbie, S.E., Chapin, S.F.I., 1996. Winter regulation of tundra litter carbon and nitrogen dynamics. *Biogeochemistry* 35, 327–338.
- Homyak, P.M., 2006. *Nitrogen Immobilization by Woodchip Application: Protecting Water Quality in a Northern Hardwood Forest*. M.S. Thesis. SUNY College of Environmental Science and Forestry, Syracuse, NY.
- Hornbeck, J.W., Kropelin, W., 1982. Nutrient removal and leaching from a whole-tree harvest of northern hardwoods. *J. Environ. Qual.* 11, 309–316.
- Hornbeck, J.W., Martin, C.W., Pierce, R.S., Bormann, F.H., Likens, G.E., Eaton, J.S., 1986. Clearcutting northern hardwoods: effects of hydrologic and nutrient ion budgets. *Forest Sci.* 32, 667–686.
- Idol, T.W., Pope, P.E., Ponder, F.J., 2003. N mineralization, nitrification, and N uptake across a 100-year chronosequence of upland hardwood forests. *Forest Ecol. Manage.* 176, 509–518.
- Likens, G.E., Bormann, F.H. (Eds.), 1995. *Biogeochemistry of a Forested Ecosystem*. Springer-Verlag, New York.
- Lovett, G.M., Rueth, H., 1999. Soil nitrogen transformations in beech and maple stands along a nitrogen deposition gradient. *Ecol. Appl.* 9, 1330–1344.
- Lovett, G.M., Weathers, K.C., Sobczak, W.V., 2000. Nitrogen saturation and retention in forested watersheds of the Catskill Mountains, New York. *Ecol. Appl.* 10, 73–84.
- Martin, C.W., 1988. Soil disturbance by logging in New England—review and management recommendations. *North. J. Appl. Forestry* 5, 30–34.
- Martin, C.W., Hornbeck, J.W., 1994. Logging in New England need not cause sedimentation of streams. *North. J. Appl. Forestry* 11, 17–23.
- Martin, C.W., Hornbeck, J.W., Likens, G.E., Buso, D.C., 2000. Impacts of intensive harvesting on hydrology and nutrient dynamics of northern hardwood forests. *Can. J. Fisher. Aquat. Sci.* 57, 19–29.
- Mazzarino, M.J., Szott, L., Jimenez, M., 1993. Dynamics of soil total C and N, microbial biomass, and water-soluble C in tropical agroecosystems. *Soil Biol. Biochem.* 25, 205–214.
- Messina, M.G., Schoenholtz, S.H., Lowe, M.W., Wang, Z., Gunter, D.K., Londo, A.J., 1997. Initial responses of woody vegetation, water quality, and soils to harvesting intensity in a Texas bottomland hardwood ecosystem. *Forest Ecol. Manage.* 90, 201–215.
- Micks, P., Downs, M.R., Magill, A.H., Nadelhoffer, K.J., Aber, J.D., 2004. Decomposing litter as a sink for ^{15}N -enriched additions to an oak forest and a red pine plantation. *Forest Ecol. Manage.* 196, 71–87.
- Murdoch, P.S., Stoddard, J.L., 1992. The role of nitrate in the acidification of streams in the Catskill Mountains of New York. *Water Resour. Res.* 28, 2707–2720.
- Murdoch, P.S., Stoddard, J.L., 1993. Chemical characteristics and temporal trends in eight streams of the Catskill Mountains, New York. *Water Air Soil Pollut.* 67, 367–395.

- NADP, 2006. Inorganic nitrogen wet deposition from nitrate and ammonium [Online] <http://nadp.sws.uiuc.edu> (verified 4-24-06).
- NRCC, 2005. CLIMOD [Online]. Available by Cornell University <http://climod.nrcc.cornell.edu/> (posted March 31, 2006; verified March 31, 2006).
- Nyland, R.D., 2002. *Silviculture: Concepts and Applications*, Second ed. McGraw-Hill, New York.
- Ohle, N., Sebestyen, S.D., Shanley, J.B., Doctor, D.H., Kendall, C., Wankel, S.D., Boyer E.W., 2004. Tracing sources of nitrate in snowmelt runoff using a high-resolution isotopic technique. *Geophys. Res. Lett.* 31, L21506, doi:10.1029/2004GL020908.
- Ollinger, S.V., Aber, J.D., Lovett, G.M., Millham, S.E., Lathrop, R.G., Ellis, J.M., 1993. A spatial model for atmospheric deposition for the northeastern U.S. *Ecol. Appl.* 3, 459–472.
- Park, B.B., Yanai, R.D., Germain, R.H. 2006. Effects of silvicultural treatments on carbon and nutrient storage of coarse woody debris in the Catskill region. The Ecological Society of America 91st annual meeting abstracts.
- Paustian, K., Schnürer, J., 1987. Fungal growth response to carbon and nitrogen limitation: application of a model to laboratory and field data. *Soil Biol. Biochem.* 19, 621–629.
- Schimel, J.P., Hättenschwiler, S., 2007. Nitrogen transfer between decomposing leaves of different N status. *Soil Biol. Biochem.* 39, 1428–1436.
- Schipper, L.A., Vojvodic-Vukovic, M., 1998. Nitrate removal from groundwater using a denitrification wall amended with sawdust: field trial. *J. Environ. Qual.* 27, 664–668.
- Setälä, H., Marshall, V., Trofymow, J., 1996. Influence of body size of soil fauna on litter decomposition and ¹⁵N uptake by poplar in a pot trial. *Soil Biol. Biochem.* 28, 1661–1675.
- Skroch, W.A., Powell, M.A., Bilderback, T.E., Henry, P.H., 1992. Mulches: Durability, aesthetic value, weed control, and temperature. *J. Environ. Hortic.* 10, 43–45.
- Sobczak, W.V., Findlay, S.E., Dye, S., 2003. Relationships between DOC bioavailability and nitrate removal in an upland stream: an experimental approach. *Biogeochemistry* 62, 309–327.
- Takahashi, M., Ikeda, S., 1994. Effects of clearfelling on the mineral nitrogen dynamics in the surface soil of a Japanese cedar (*Cryptomeria japonica* D. Don) plantation —A comparison between slash-removal and slash-scattering treatments. *J. Jpn. Forest Soc.* 76, 224–232.
- Torns, L.A., 1979. *Soil Survey of Ulster County, New York*. US Department of Agriculture, Soil Conservation Service.
- USDA, N.R.C.S. 2006. Keys to soil taxonomy [Online] http://soils.usda.gov/technical/classification/tax_keys/keys.pdf (posted July 21, 2006; verified October 20, 2006).
- Venterea, R.T., Lovett, G.M., Groffman, P.M., Schwarz, P.A., 2003. Landscape patterns of net nitrification in a northern hardwood-conifer forest. *Soil Sci. Soc. Am. J.* 67, 527–539.
- Waldrop, M.P., McColl, J.G., Powers, R.F., 2003. Effects of forest postharvest management practices on enzyme activities in decomposing litter. *Soil Sci. Soc. Am. J.* 67, 1250–1256.
- Wang, X., Burns, D.A., Yanai, R.D., Briggs, R.D., Germain, R.H., 2006. Changes in stream chemistry and nutrient export following a partial harvest in the Catskill Mountains, New York, USA. *Forest Ecol. Manage.* 223, 103–112.
- Wheeler, G.L., Steele, K.F., Lawson, E.R., 2000. Water and nutrient movement in small, forested watersheds in the Boston Mountains of Arkansas. *Forest Sci.* 46, 335–343.
- Wyland, L.J., Jackson, L.E., 1993. Evaluating nitrate recovery by ion-exchange resin bags. *Soil Sci. Soc. Am. J.* 57, 1208–1211.
- Zeller, B., Colin-Belgrand, M., Dambrine, E., Martin, F., Bottner, P., 2000. Decomposition of ¹⁵N-labelled beech litter and fate of nitrogen derived from litter in a beech forest. *Oecologia* 123, 550–559.