

ANALYSIS OF NITROGEN DYNAMICS IN THE LYE BROOK WILDERNESS AREA, VERMONT, USA

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Abstract. Nitrogen (N) deposition and its impact on terrestrial and aquatic ecosystems is a concern facing federal land managers at the Lye Brook Wilderness in Vermont and other protected areas throughout the northeastern United States. In this study, we compared N production in soils with N concentrations and outputs in leachates to determine how forest cover types differ in regulating N losses. Also, precipitation inputs and modeled estimates of streamwater outputs were used to calculate a watershed N budget. Most ammonium and nitrate were produced in organic soils with deciduous cover. Softwood stands had low net nitrification rates and minimal N leaching. A comparison of watershed inputs and outputs showed a net gain in total dissolved N ($5.5 \text{ kg ha}^{-1} \text{ yr}^{-1}$) due to an accumulation of dissolved inorganic N. The Lye Brook Wilderness ecosystem has N budgets similar to other forested ecosystems in the region, and appears to be assimilating the accumulating N. However seasonal losses of nitrate observed in mineral soils and streamwater may be early warnings of the initial stages of N saturation.

Keywords: nitrification, nitrogen, nitrogen mineralization, soil water, stream chemistry, wilderness area

1. Introduction

High inputs of nitrogen (N) have raised concerns about N saturation in forests of the northeastern United States. Throughout the region, undisturbed forests are typically N-limited and some forests have a tremendous capacity to retain additional N in biomass and soils (Aber *et al.*, 1995). Other forested ecosystems are more limited in this capacity and may reach a point where N inputs exceed the biological demand for N (Kahl *et al.*, 1993; Norton *et al.*, 1994; Gilliam *et al.*, 1996). Forests that are exposed to elevated N additions may exhibit several characteristic responses, including plant nutrient imbalances (Schulze, 1989; Aber *et al.*, 1995; Cronan and Grigal, 1995), cation depletion (Reuss and Johnson, 1986), and acidification and eutrophication of streamwater due to additions and production of $\text{NO}_3\text{-N}$ (Henriksen *et al.*, 1992; Stoddard, 1994).

Fenn *et al.* (1998) summarized forest management practices aimed at ameliorating the effects of high N deposition. These strategies include removing N in biomass through harvesting or burning, altering stand composition to reduce N-fixing species, and enhancing denitrification through riparian zone management. These options may be effective in forests that are intensely managed, but in some protected areas such activities are not permitted. In this study we investigate issues surrounding N saturation in the Lye Brook Wilderness (LBW) in southeastern Vermont, USA. The wilderness designation poses a challenge to land managers; it charges them with preserving the wilderness characteristics of the site, such as water quality, vegetation and soils, but eliminates the aforementioned options for removing N from the ecosystem.

The only option available to wilderness area managers to protect forests from increased N deposition is limiting emissions of N. This is accomplished through a permitting process pursuant to the Clean Air Act Amendments of 1977. This process requires permits for new sources of pollution or modifications of existing sources that may have an impact on wilderness areas such as the LBW. In order to evaluate permit applications, it is necessary to assess the effects of N deposition, and to estimate the sensitivity of ecosystems to deposition changes (Adams *et al.*, 1991). The purpose of this study was to determine the N status of the LBW by examining N production and retention at the forest stand and watershed scale. This analysis was used to determine whether the ecosystem is accumulating N and to gain insight as to the potential for N saturation. The analysis, which relies on a relatively low intensity of field data collection, also serves as an example for examining N cycling in other protected areas.

2. Methods

2.1. SITE DESCRIPTION

The 6,280-ha LBW is located within the boundary of the Green Mountain National Forest in southwestern Vermont, USA. Combined wet and dry deposition of N is approximately 9 to 11 kg ha⁻¹ yr⁻¹ (Ollinger *et al.*, 1995), annual precipitation in the region averages 1150 mm, and the mean annual temperature is 7 °C. In this unreplicated case study, a 130-ha watershed (43°07'N, 73°02'W), draining a perennial, first-order stream, was chosen to investigate N cycling within the wilderness. The watershed ranges in elevation from 725 to 850 m and has an average slope of 7%. Underlying bedrock is of the Mount Holly Complex and

Mendon Formation composed primarily of quartzite, schist, amphibolite and gneiss. Soils consist of Typic Humaquepts, Epiaquods, and Haplorthods that have formed in compact, loamy, glacial till. Most of the watershed was logged between 1915 and 1920, resulting in an even-aged forest with a forest cover of 51% hardwood, 29% softwood and 20% mixed hardwood and softwood types. Conifer species consist almost entirely of red spruce (*Picea rubens*) and balsam fir (*Abies balsamea*) and deciduous species include sugar maple (*Acer saccharum*), red maple (*Acer rubrum*), paper birch (*Betula papyrifera*), yellow birch (*Betula alleghaniensis*), and American beech (*Fagus grandifolia*).

2.2. ECOSYSTEM PRODUCTION OF N

Net N mineralization and net nitrification rates were measured by *in situ* incubation in buried bags from June 1994 through October 1995. Details from the method are given in Eno (1960) and Federer (1983). Three soil pits were excavated at each of three unreplicated sites consisting of softwood, mixed hardwood and softwood, and hardwood cover types. The sites were located 100 to 250 m apart and were approximately 1.2 km southeast of the watershed boundary in stands that were representative of the watershed. At each soil pit, a polyethylene bag was filled with representative soil and buried in the Oa and B horizon. Soil bags were incubated at 28-day intervals from May through November, and were left in the ground from December through April to measure rates of net N mineralization and net nitrification during winter. An analysis of variance was conducted to examine soil-bag net N mineralization and net nitrification results among sites and between horizons. Where results were significant, a Bonferroni multiple-comparison test was used to determine levels of significance between means.

Net N mineralization and net nitrification rates were converted from units of mass ($\mu\text{gN g soil}^{-1}$) to an areal basis (kg N ha^{-1}) using bulk density, horizon thickness, and percent rock fraction. Soil bulk density was estimated from the organic matter fraction based on the inverse relationship found in forest soils of New England (Federer *et al.*, 1993). Because the percent rock fraction was not measured, ranges for net N mineralization and net nitrification were calculated based on rock fractions of 0 to 30% which is typical for this region. Net N mineralization and net nitrification rates were weighted for the entire watershed by multiplying the rate at each site times the percent hardwood (51%), softwood (29%) and mixed (20%) cover types for the watershed.

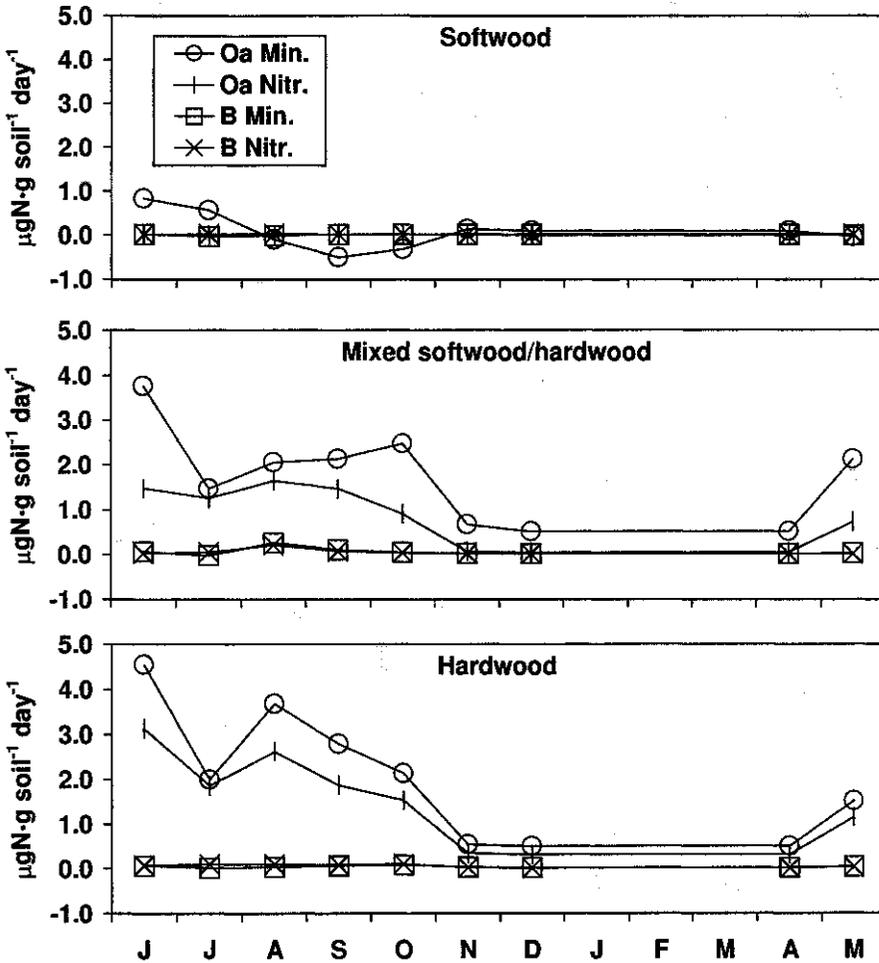


Fig. 1. Mean net N mineralization and net nitrification rates. Sampling period is June, 1994 through May, 1995.

2.3. WATER SAMPLES

Streamwater sampling was conducted bi-weekly from May 26, 1994 to August 30, 1995 and every three weeks from May 15 to October 18, 1996. In November 1994, three tension lysimeters were installed in the Oa and B soil horizons at each of the three sites where net N mineralization and net nitrification rates were measured. Soil water samples were collected bi-weekly from June 6 to October 4, 1995 and every three weeks from May 15 to October 18, 1996. Measurements of ammonium ($\text{NH}_4\text{-N}$) and nitrate ($\text{NO}_3\text{-N}$) in soil water and streamwater were made using continuous flow colorimetry. Total dissolved nitrogen (TDN) was measured using an Antek N detector and dissolved organic

nitrogen (DON) was calculated by subtracting dissolved inorganic nitrogen (DIN: $\text{NH}_4\text{-N} + \text{NO}_3\text{-N}$) from TDN (Merriam *et al.*, 1996). An analysis of variance was used to compare differences among sites and horizons for all soil water chemistry measurements, which is a similar approach used to analyze buried bag data described previously.

To calculate stream and soil water fluxes, estimates of hydrologic fluxes were made using the Brook90 model (Federer, 1998). This model has been used extensively to compute water budgets for small, forested watersheds in New England (Federer and Lash, 1978; Hornbeck *et al.*, 1986). Brook90 simulates water movement through multiple soil layers, making it possible to approximate the flux of soil water from different horizons, in addition to streamwater. The model requires numerous site-specific parameters related to catchment, canopy, and soil. Values for these parameters were based on field observations and available literature. Brook90 was run on a daily timestep using daily precipitation and minimum and maximum temperatures collected at the closest National Weather Service station in Dorset, Vermont, approximately 10 km from the sampling area at an elevation of 284 m. Precipitation and temperature data were scaled for elevational trends using multiple regression equations (Ollinger *et al.*, 1995).

Solute fluxes were calculated by multiplying the stream or soil water concentration (mg L^{-1}) for each sampling date by the daily water flux ($\text{L ha}^{-1} \text{ day}^{-1}$) surrounding each chemical sample. Lysimeter samples were not collected during the winter months; therefore soil water flux for this period was based on concentrations from the last sample collected in the fall to the first sample collected in the spring. Relatively few lysimeter samples were collected from the Oa horizon of the hardwood stand because the organic soil at this site was nearly always too dry to extract a sample. Therefore, the watershed flux calculation for the Oa horizon only includes data from the mixed and softwood sites.

Atmospheric inputs of $\text{NH}_4\text{-N}$ and $\text{NO}_3\text{-N}$ were calculated using weekly chemical data collected at the Bennington, Vermont NADP/NTN station located approximately 50 km south of the site (National Atmospheric Deposition Program, Colorado State University). The sampling period used for atmospheric chemistry was the same as the stream sampling period (May 26, 1994 to August 30, 1995; May 15 to October 18, 1996). Since NADP/NTN data do not include organic N measurements, concentrations of DON in precipitation were obtained from the Hubbard Brook Experimental Forest in New Hampshire. In this study, dry N deposition was not included in the budget calculations. Predicted dry N deposition in the northeastern United

Table I. Average annual soil water concentrations (mg L^{-1}) of $\text{NH}_4\text{-N}$, $\text{NO}_3\text{-N}$ and DON for softwood, mixed softwood and hardwood, and hardwood sites.

	Oa Horizon			B Horizon		
	$\text{NH}_4\text{-N}$	$\text{NO}_3\text{-N}$	DON	$\text{NH}_4\text{-N}$	$\text{NO}_3\text{-N}$	DON
Softwood						
Mean	0.67	0.03	0.59	0.05	0.16	0.25
Std. Dev.	0.22	0.04	0.28	0.03	0.06	0.05
N	26	26	13	41	41	19
Mixed						
Mean	0.60	0.27	1.06	0.06	0.78	0.48
Std. Dev.	0.24	0.41	0.13	0.03	0.46	0.18
N	23	23	13	41	41	19
Hardwood						
Mean	0.58	0.16	0.25	0.05	0.32	0.34
Std. Dev.	0.28	0.25	—	0.03	0.29	0.07
N	5	4	1	38	38	20

States ranges from 46% of wet deposition in southwestern Pennsylvania to 20% of wet deposition in northern Maine (Ollinger *et al.*, 1995).

3. Results

3.1. NET N MINERALIZATION AND NET NITRIFICATION

Most net N mineralization and net nitrification occurred in the Oa or organic soil horizons at all sites and very low rates were measured in B or mineral soil horizons (Figure 1). Average annual net N mineralization in the Oa horizon was highest at the hardwood site ($1.64 \mu\text{gN g soil}^{-1} \text{ day}^{-1}$), followed by the mixed site ($1.43 \mu\text{gN g soil}^{-1} \text{ day}^{-1}$) and the softwood site ($0.09 \mu\text{gN g soil}^{-1} \text{ day}^{-1}$), but there was no statistical difference between the hardwood and mixed site at the $P=0.05$ level. Net nitrification in the Oa horizon followed a similar pattern with significantly higher average annual rates occurring at the hardwood ($1.17 \mu\text{gN g soil}^{-1} \text{ day}^{-1}$) and mixed sites ($0.64 \mu\text{gN g soil}^{-1} \text{ day}^{-1}$), and rates close to zero at the softwood site ($<0.01 \mu\text{gN g soil}^{-1} \text{ day}^{-1}$).

Seasonal patterns for net N mineralization and net nitrification in organic soils were observed at all sites (Figure 1). At the hardwood and mixed sites, rates were higher from May through October and were lower during the rest of the year. The softwood site had higher net N mineralization in the Oa horizon during June and July and rates that

Table II. Average annual N budgets ($\text{kg ha}^{-1} \text{ yr}^{-1}$). Net values equal input in precipitation (Ppt) minus output in soil water from the Oa and B horizons, and streamwater. DIN budgets for HB, CP and SR are from Hornbeck *et al.* (1997) and budgets for BNA are from Martin (1979).

	LBW					HB	CP	SR	BNA
	NH ₄ -N	NO ₃ -N	DON	TDN	DIN	DIN	DIN	DIN	DIN
Input									
Ppt	2.4	4.8	1.3	8.5	7.2	7.6	6.9	6.4	8.8
N min./Nitr.									
Oa	58-83	35-50	—	—	—	—	—	—	—
B	10-15	12-17	—	—	—	—	—	—	—
Output									
Oa	3.3	0.5	4.1	7.9	3.8	—	—	—	—
B	0.4	2.5	2.2	5.1	2.9	—	—	—	—
Stream	0.1	0.8	2.1	3.0	0.9	0.6	0.2	1.4	7.0
Net (input minus output)									
Oa	-0.9	4.3	-2.8	0.6	3.4	—	—	—	—
B	2.0	2.3	-0.9	3.4	4.3	—	—	—	—
Watershed	2.3	4.0	-0.8	5.5	6.3	7.0	6.7	5.0	1.8

were less than zero during August through October, indicating that immobilization occurred during this time. Net nitrification rates in the Oa horizon of the softwood site were low throughout the year and did not exhibit strong seasonal effects.

3.2. SOIL WATER AND STREAMWATER CHEMISTRY

Soil water concentrations of NH₄-N were similar at all sites (Table I), but were significantly higher in Oa than in B horizons ($P < 0.001$). Concentrations of NO₃-N in organic horizons were lowest at the softwood site (0.03 mg L^{-1}), intermediate at the hardwood (0.16 mg L^{-1}) site, and highest at the mixed site (0.27 mg L^{-1}). This same pattern for NO₃-N occurred in mineral soils, but concentrations were significantly higher in the B than in the Oa horizons ($P < 0.001$). Concentrations of DON in the Oa horizon were also highest at the mixed site (1.06 mg L^{-1}) with lower concentrations occurring at the softwood (0.59 mg L^{-1}) and hardwood site (0.25 mg L^{-1}). However, the DON value at the hardwood site is based on one sample.

Seasonal trends were evident for some of the measured solutes in soil water and streams (Figure 2). In Oa soil solution, NH₄-N, NO₃-N

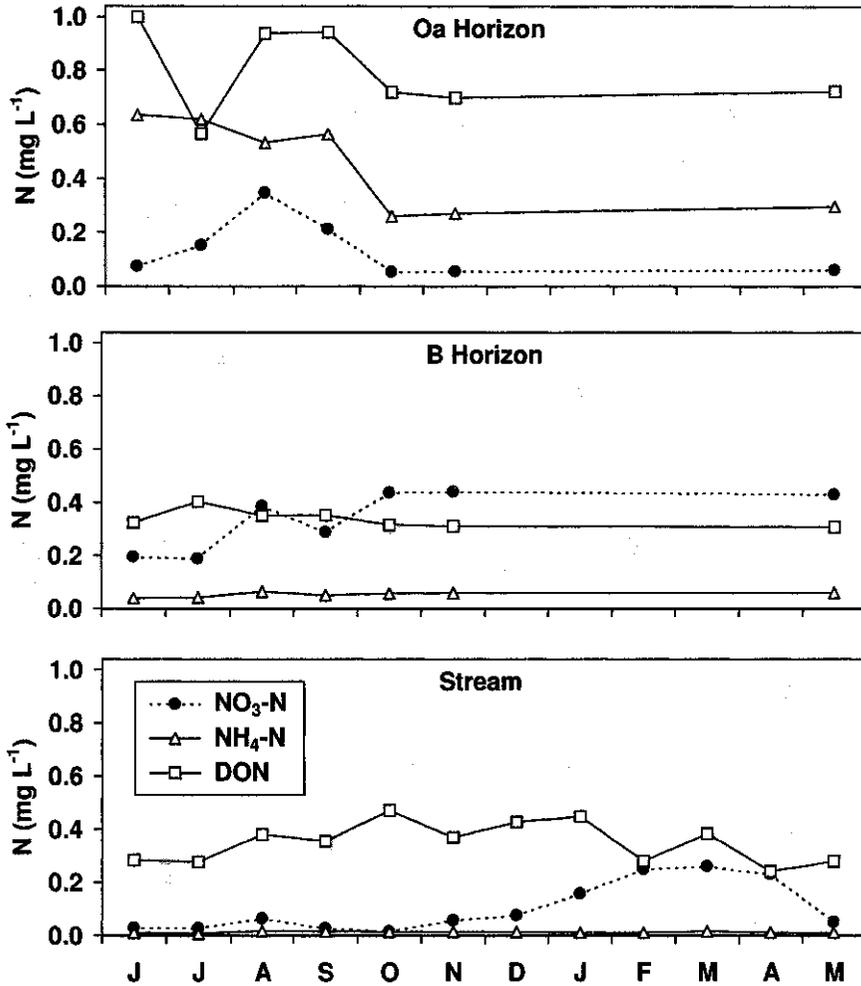


Fig. 2. Mean monthly concentrations of NO₃-N, NH₄-N, and DON (mg L⁻¹) in soil water of the Oa and B horizons and streamwater.

and DON were highest during the summer, usually peaking in August. In the B horizon, NO₃-N was lower from June through September while other measured solutes remained relatively constant throughout the year. In streamwater, the only form of N exhibiting a seasonal pattern was NO₃-N, which had low concentrations during the growing season, followed by an increase in winter, reaching the highest level in March.

3.3. NITROGEN BUDGETS

Annual N input in precipitation was dominated by DIN, with NO₃-N and NH₄-N accounting for 85% of total inputs (Table II). In soils,

production of $\text{NH}_4\text{-N}$ by net N mineralization was greatest in the Oa horizon. More $\text{NO}_3\text{-N}$ was also produced in organic soils than in mineral soils. Soil leachates draining the Oa horizon were dominated by $\text{NH}_4\text{-N}$ with lesser amounts of $\text{NO}_3\text{-N}$, but flux from the B horizon was dominated by $\text{NO}_3\text{-N}$ with lesser amounts of $\text{NH}_4\text{-N}$. Streamwater outputs of DIN were dominated by $\text{NO}_3\text{-N}$ and $\text{NH}_4\text{-N}$ outputs were low. The greatest DON flux occurred from the Oa horizon and decreased with depth, resulting in a lower streamwater flux. TDN values showed that for the period of the study, the watershed accumulated N at a rate of $5.5 \text{ kg ha}^{-1} \text{ yr}^{-1}$, due exclusively to a net gain in DIN.

4. Discussion

The LBW presents an interesting dilemma to land managers because of mandates to protect the area against anthropogenic N deposition. In forests that are intensely managed, practices such as harvesting and burning result in continued removal of N from the ecosystem. These activities also create biomass sinks for N because aggrading forests require more N for growth (Vitousek and Reiners, 1975). The elimination of forest harvesting as a management option results in mature forests with higher N capitals, that may be more vulnerable to N saturation. Thus it is important to understand the processes and recognize changes that may signal N saturation.

Elevated $\text{NO}_3\text{-N}$ leaching is considered an indicator of excess N in forested ecosystems. The amount of $\text{NO}_3\text{-N}$ in soil leachates and streamwater is dependent in part, on net N mineralization and net nitrification rates (Aber *et al.*, 1989; Stoddard, 1994). These processes vary with forest cover type, so an understanding of differences among stands is important in assessing the N-status of sites. Our results are consistent with other studies that have compared net N mineralization and net nitrification rates at sites with different cover types (Nadelhoffer *et al.* 1985; Hill and Shackleton, 1989). Like these studies, we found low net N mineralization and almost no net nitrification in conifer stands, and higher rates in deciduous stands. These differences have been attributed to differences in soil moisture, temperature, pH, as well as biotic controls related to competition between plants and microbes (Robertson, 1982; Hill and Shackleton, 1989; Aber *et al.*, 1993).

Differences in net N mineralization and net nitrification among stands can be compared to N concentrations in soil water to clarify relationships between production and losses of N in soil leachates. In the softwood stand, low rates of net nitrification resulted in lower $\text{NO}_3\text{-N}$ concentrations in leachates (Figure 1, Table I). The highest

NO₃-N concentrations in soil water were found at the mixed site which was unexpected because net nitrification rates were lower compared to the hardwood site (for the hardwood stand this comparison was based solely on B horizon leachates since only 5 samples were collected from the Oa horizon due to dry soil conditions). This suggests that factors in addition to nitrification may be important in regulating NO₃-N concentrations in soil water, such as greater uptake of N by deciduous species (Nadelhoffer *et al.*, 1995). Concentrations of NH₄-N in soil water were nearly equal at all sites and it appears that although net N mineralization rates were lower in the softwood stand, less NH₄-N was being nitrified, resulting in similar NH₄-N concentrations among sites.

Leaching losses at the softwood site were low which indicates that at current deposition levels, conifer stands retain more N than mixed and hardwood stands. However, this may not mean that softwood stands will be able to retain more N if deposition increases. Hardwood species are a better sink for N than softwood species because hardwoods assimilate more N in foliage and bolewood (Nadelhoffer *et al.*, 1995). If softwood sites have lower rates of N assimilation, they may be more sensitive to increases in N deposition. Softwood stands also have a greater capacity to trap dry N deposition because softwood species have a greater leaf area index and because softwood needles are retained throughout the year (Lovett *et al.*, 1996). Although dry N deposition was not measured in this study, we expect that inputs would be higher in softwood stands resulting in greater N retention.

Concentrations of NO₃-N in streamwater, which serves as an indicator of how well N is assimilated by forests, averaged 0.1 mg L⁻¹ for the study period. A summary by Hornbeck *et al.* (1997) of NO₃-N concentrations in streams of 159 forested watershed in central New England showed a range of 0 to 0.8 mg L⁻¹. Forty-eight percent of the 159 watersheds had higher streamwater NO₃-N concentrations than the LBW suggesting that forest on LBW are not losing excessive amounts of inorganic N.

Annual N budgets show that the watershed investigated in this study accumulated N at a rate of approximately 5.5 kg ha⁻¹ yr⁻¹, due primarily to net gains of NO₃-N, and NH₄-N (Table II). There was a slight net loss in DON (-0.8 kg ha⁻¹ yr⁻¹), however this did not offset the gain in DIN. Approximately 60% of the NH₄-N produced in the Oa horizon was nitrified. The low flux of NO₃-N from organic soil horizons suggests that much of this NO₃-N was taken up by vegetation. Low NO₃-N concentrations in soil water and streams during growing season months also support the idea of uptake by vegetation.

Table II compares DIN budgets at LBW to those published for the Bowl Natural Area (BNA) in New Hampshire (Martin, 1979) and the Hubbard Brook Experimental Forest (HB) and Cone Pond Watershed (CP) in New Hampshire and the Sleeper's River Watershed (SR) in Vermont (Hornbeck *et al.*, 1997). N retention among the sites ranged from a low at BNA ($1.8 \text{ kg ha}^{-1} \text{ yr}^{-1}$) to high at HB ($7.0 \text{ kg ha}^{-1} \text{ yr}^{-1}$). The low retention at BNA was suggested to be the result of inefficient N cycling by the old-growth forest that covers the watershed (Vitousek and Reiners, 1975). The net gain in DIN at LBW ($6.3 \text{ kg ha}^{-1} \text{ yr}^{-1}$) was intermediate in comparison to HB, CP, and SR, suggesting that forests at LBW are cycling N in a manner similar to other second-growth forests in the region.

In mineral soils at the LBW, net nitrification rates were low, but the flux of $\text{NO}_3\text{-N}$ from mineral horizons was high ($2.5 \text{ kg ha}^{-1} \text{ yr}^{-1}$) compared to organic soils ($0.5 \text{ kg ha}^{-1} \text{ yr}^{-1}$) (Table II). Seasonal patterns of $\text{NO}_3\text{-N}$ in the B horizon showed that concentrations of $\text{NO}_3\text{-N}$ remained low throughout the summer, and that concentrations were higher during the winter (Figure 2). A similar pattern was evident for streamwater, but $\text{NO}_3\text{-N}$ outputs in streams were low compared to mineral soil outputs. The high $\text{NO}_3\text{-N}$ values in mineral soil and low values in streamwater suggest that $\text{NO}_3\text{-N}$ was either accumulating in mineral soil, being lost through denitrification or uptake in the riparian zone, or was being utilized within the stream. If the difference between the flux of $\text{NO}_3\text{-N}$ in B horizon soil water and streamwater was due to losses in the stream or riparian zone, the watershed may have accumulated less N. These potential pathways for N loss were not measured in this study, but should be considered in future research. Whatever the mechanism, annual $\text{NO}_3\text{-N}$ losses in streamwater were low compared to inputs of N, indicating that the watershed is a sink for N at current deposition levels.

5. Conclusions

While our study involved a relatively low level of ecosystem measurements, we were able to show that the LBW is accumulating N, and that the ecosystem has not yet exceeded its capacity to assimilate N. $\text{NO}_3\text{-N}$ in streamwater was low, although seasonal increases, and the possible accumulation of $\text{NO}_3\text{-N}$ in mineral soil, may be warning signals of the initial stages of N saturation. Without the benefit of long-term data, it is difficult to determine if additional N inputs will cause an increase in $\text{NO}_3\text{-N}$ leaching. Prior to making this determination, it is essential to establish more detailed baseline levels for N concentrations and budgets

and other biogeochemical parameters such as pH and base cations that may be affected by increases in N deposition. Monitoring N retention at the watershed level is an important first step in determining the N status of sites and providing land managers with data to determine if restrictions on N emissions are warranted.

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