Forest N Dynamics after 25 years of Whole Watershed N Enrichment: The Bear Brook Watershed in Maine

Article *in* Soil Science Society of America Journal · February 2019

DOI: 10.2136/sssaj2018.09.0348

6 authors, including:

Kaizad Patel
Pacific Northwest National Laboratory

Ivan J Fernandez
University of Maine

Sarah J. Nelson
Appalachian Mountain Club

Aaron Robert Weiskittel
University of Maine

Published online May 9, 2019

North American Forest Soils Conference - International Symposium on Forest Soils

Forest N Dynamics after 25 years of Whole Watershed N Enrichment: The Bear Brook Watershed in Maine

Kaizad F. Patel*
School of Forest Resources
Univ. of Maine
5755 Nutting Hall
Orono, ME 04469

Ivan J. Fernandez
School of Forest Resources
Univ. of Maine
5755 Nutting Hall
Orono, ME 04469
and
Climate Change Inst.
Univ. of Maine
5764 Sawyer Res. Center
Orono, ME 04469

Sarah J. Nelson
School of Forest Resources
Univ. of Maine
5755 Nutting Hall
Orono, ME 04469

Marie-Cécile Gruselle
School of Forest Resources
Univ. of Maine
5755 Nutting Hall
Orono, ME 04469
and
Friedrich-Schiller Univ. Inst. of Geography
Dep. of Soil Science
Löbdergraben 32
07743 Jena
Germany

Stephen A. Norton
Climate Change Inst.
Univ. of Maine
5764 Sawyer Res. Center
Orono, ME 04469
and
School of Earth and Climate Sciences
Univ. of Maine
5790 Bryand Global Sciences Center
Orono, ME 04469

Aaron R. Weiskittel
School of Forest Resources
Univ. of Maine
5755 Nutting Hall
Orono, ME 04469

We examine the temporal trend of input–output N fluxes and net ecosystem N retention, and estimate a mass balance for ecosystem soil and vegetation pools, after 25 yr of chemical manipulation at the Bear Brook Watershed in Maine (BBWM). The BBWM is a paired whole watershed manipulation experiment designed to study the effects of elevated N and S deposition on forest ecosystem function. Starting in 1989, 25.2 kg N ha^{-1} was added annually to the West Bear (treated) watershed. The N additions in West Bear stimulated N loss through stream exports, and West Bear retained 81% of the annual N inputs, compared to 94% retention in the reference, East Bear. After 25 yr of N additions, the West Bear watershed had accumulated ~700 kg ha^{-1} more N than East Bear in soils and vegetation, with ~10% of the accumulated N stored in forest biomass. The treatment increased recent biomass N accumulation rates in the hardwood stands, but not in the softwoods. Soils did not show detectable differences in total N content between watersheds, although the organic soils had greater N in West Bear. This paper presents a unique set of findings from one of the few long-term whole forest ecosystem N enrichment studies in the world. While N dynamics were clearly altered in West Bear, with evidence of accelerated N cycling, the treated watershed did not attain an advanced stage of N saturation during the study period, based on the evidence from forest growth and stream N exports.

Abbreviations: BBWM, Bear Brook Watershed in Maine; DIN, dissolved inorganic nitrogen; DON, dissolved organic nitrogen.

Temperate forest productivity is typically nitrogen (N)-limited (Fenn et al., 1998; Thomas et al., 2010). Many forests in North America and Europe have shown increased growth rates in response to elevated N deposition (Thomas et al., 2010). Some ecosystems, however, have exceeded the critical loads beyond which negative effects occur (McNulty et al., 2007; Pardo et al., 2011; Tietema et al., 1998). Environmental monitoring and experimental manipulation studies show that N-enriched ecosystems have increased N losses via leaching (Lovett and Goodale, 2011; Lu et al., 2011; Templer et al., 2012a; Niu et al., 2016) and denitrification (Morse et al., 2015; Templer et al., 2012b), increased tree stress and fine root mortality (Ferretti et al., 2015; Gundersen et al., 1998; Minocha et al., 2015; Smithwick et al., 2013), and altered microbial community structure and

Core Ideas

- **Chronic N enrichment increased N exports, although ecosystem N retention was still high.**
- **Soil was the largest ecosystem N pool, but it was not significantly altered by chronic N enrichment.**
- **Chronic N enrichment increased biomass N accumulation in hardwood but not in softwood stands.**

Soil Sci. Soc. Am. J.
doi:10.2136/sssaj2018.09.0348
Received 25 Sept. 2018.
Accepted 11 Feb. 2019.
*Corresponding author (kaizad.patel@maine.edu).

function (Carrara et al., 2018; Kopáček et al., 2013; Morrison et al., 2018; Treseder, 2008). The responses varied with the magnitude of N inputs, vegetation type, disturbance history, geographic location, climate, and the duration of the experiments.

Small watersheds are useful as ecological observatories, and long-term watershed studies allow us to understand complex and interactive ecological responses. Multi-decadal monitoring of N processes in the United States and Europe has demonstrated that short-term effects may not scale up to longer time scales. Studies at the Hubbard Brook Experimental Forest in New Hampshire, USA, revealed changes in N source–sink relationships between the 1970s and the early 2000s (Yanai et al., 2013). Multi-decadal N addition experiments at the Harvard Forest (Massachusetts, USA) and Mount Ascutney (Vermont, USA) documented accelerated growth of red spruce (*Picea rubens* Sarg.) trees in the initial years of treatment, followed by declines in productivity and increased mortality in the later years (Magill et al., 2004; McNulty et al., 2017). In the NITREX experiment in Gårdsjön, Sweden, N additions induced stream nitrate (NO_3^-) leaching, although isotopic analysis indicated that the source of the leached N changed with time. Initially, leached N originated from the added N, but in the later years, the leached N originated from nitrification of the 'old' N present in the ecosystem prior to the manipulation (Moldan and Wright, 2011). Nave et al. (2009) reported significant temporal lags in soil response to N additions; mineral soils showed significant increases in C and N pools only after 15 to 20 yr of treatment. Climate variability and stochastic events increase nitrification rates and stream NO_3^- exports, adding to the complexity of interpreting ecosystem N processes (Casson et al., 2010; Mitchell et al., 1996). The current climate trajectory in the northeastern United States of increasing temperature and precipitation, as well as increasing climate variability (Hayhoe et al., 2007; Salinger, 2005; Wuebbles et al., 2017), can also influence N transformations and availability. There is emergent evidence of N oligotrophication in temperate forests due to increased primary production, and plant and microbial N uptake, and declining N deposition (McLauchlan et al., 2017; Craine et al., 2018; Groffman et al., 2018).

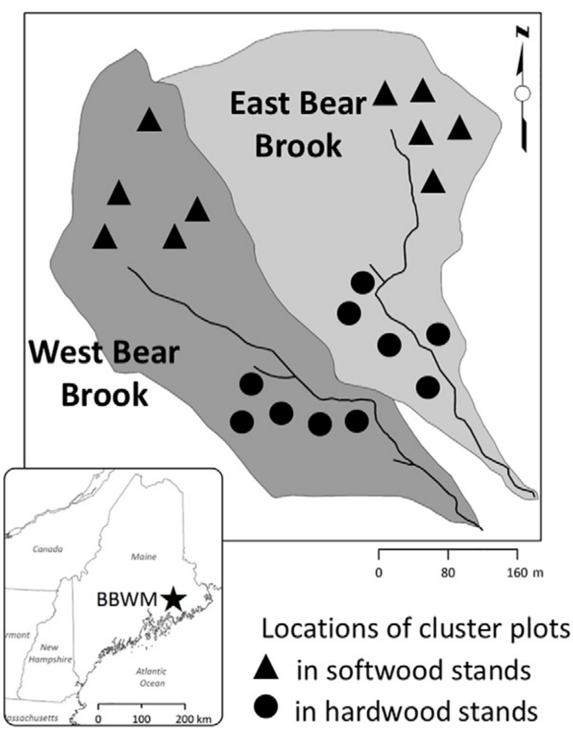

Fig. 1. Design and location (inset) of Bear Brook Watershed in Maine (BBWM), with the paired watersheds East Bear Brook (reference) and West Bear Brook (treated). Map data sources not generated by BBWM team are as follows: United States state outlines are from the National Atlas of the United States; Canadian Provinces are from the National Weather Service.

The Bear Brook Watershed in Maine (BBWM) is one of the few multi-decadal whole-ecosystem manipulation experiments in the world examining forest ecosystem response to chronic elevated N and sulfur (S) deposition. There, experimental bi-monthly additions of $(NH_4)_2SO_4$ to one watershed have increased foliar N concentrations and tree growth (Elvir et al., 2005, 2006); increased soil available N (Jefts et al., 2004; Wang and Fernandez, 1999; Patel and Fernandez, 2018); altered microbial community composition and enzyme activity (Fatemi et al., 2016; Mineau et al., 2014; Stone et al., 2012; Tatariw et al., 2018; Wallenstein et al., 2006); altered soil organic matter composition and decomposition rates (Hunt et al., 2008; Ohno et al., 2007); and increased base cation leaching (Fernandez et al., 2010) and stream NO_3^- exports (Fernandez et al., 2010; Simon et al., 2010). Given the complex and often multi-decadal nature of ecosystem N processes, it is important to understand the interactions among ecosystem components, how these may change through time, and how these changes alter N dynamics within the ecosystem.

In this paper, we (i) examine the multi-decadal trend of input–output N fluxes and net ecosystem N retention, and (ii) estimate a contemporary mass balance for ecosystem soil and vegetation pools, after 25 yr of experimental whole-ecosystem N additions at BBWM.

MATERIALS AND METHODS

Site Description

The BBWM is a long-term whole watershed acidification experiment in eastern Maine, USA (44°52′ N, 68°06′ W), established to study the effects of elevated N and S deposition on ecosystem processes (Fig. 1). The BBWM has two paired watersheds, the reference East Bear Brook (11.0 ha) and the manipulated West Bear Brook (10.3 ha), which was treated with bimonthly applications of ammonium sulfate [$(NH_4)_2SO_4$] fertilizer from above the canopy, at the rate of 28.8 kg S ha^{-1} yr^{-1} and 25.2 kg N ha^{-1} yr^{-1}, beginning in November 1989 and continuing throughout the period of study reported here (1989–2014). Both watersheds are drained by first-order streams. Bedrock consists dominantly of quartzite, phyllite, and calc-silicate low-grade metasediments, and minor granite dikes (Norton et al., 1999; SanClements et al., 2010). Soils, developed from till, are coarse-loamy, mixed, frigid Typic and Aquic Haplorthods. Average annual air temperature (2005–2014) at the site was 5.6°C (Patel et al., 2018a, 2018b), and average annual precipitation (2005–2014) was 140 cm. Vegetation was similar in both watersheds, with lower elevations dominated by hardwood species, primarily *Fagus grandifolia* Ehrh. (American beech), *Acer saccharum* Marsh. (sugar maple), *Acer rubrum* L. (red maple), and *Betula alleghaniensis* Britt. (yellow birch); higher elevations were dominated by softwood species, *Picea rubens* Sarg. (red spruce) and *Abies balsamea* L. (balsam fir) (Wang and Fernandez, 1999). The research site was divided into four compartments by N treatment and forest type: East Bear hardwoods, East Bear softwoods, West Bear hardwoods, and West Bear softwoods. Five cluster plots (10 m by 15 m) were established in each compartment.

Input–Output Nitrogen Fluxes

Precipitation and stream samples were collected during 1989 to 2014 to calculate input and output N fluxes, respectively. Wet-only precipitation was collected using an AeroChem-Metrics precipitation collector (Norton et al., 1999). Stream samples were collected as grab samples and event samples (e.g., during snowmelt and rain events) using ISCO automated samplers from both East Bear and West Bear streams to calculate dissolved inorganic N (DIN) exports. During the first half of the study period, precipitation and stream samples were collected weekly and during high stream-flow events (up to 300 samples per year). During the second half of the study period, sampling frequency was reduced, and precipitation and stream samples (>50 samples per year) were collected biweekly or monthly, and during selected hydrologic events. Measured stream chemistry from base sampling plus ISCO event samplers was interpolated between collections and coupled with hourly stream discharge to develop monthly and annual streamwater fluxes (Kahl et al., 1999; Norton et al., 2010). Precipitation and stream samples were analyzed for NH_4^+ and NO_3^- using ion chromatography at the University of Maine Sawyer Environmental Research Center. Dry deposition was not measured at the site, so for this analysis we used reported estimates of annual dry deposition from the CASTNET Howland station (HOW132), located ~60 km northwest of BBWM (CASTNET, 2018). The annual wet deposition overlapping values for the Howland station and BBWM were strongly correlated (Pearson's $r = 0.7735$, $p < 0.0001$). Annual ecosystem inorganic N retention was:

$$\text{retention, kg ha}^{-1} = \text{input} - \text{output} \quad [1]$$

$$\text{retention, \%} = (\text{input} - \text{output})/\text{input} \times 100 \quad [2]$$

where input refers to (wet + dry) inorganic N deposition and output refers to stream inorganic N exports. Ecosystem N retention values are reported in Table 1.

Organic N does not contribute significantly to the N retention mass balance at our site (see the Discussion section), and therefore inorganic N retention closely approximates total N retention at BBWM. Annual inputs, outputs, and retention were calculated for the water year from October 1 to September 30. The calculations for ecosystem inorganic N retention in Eq. [1] and [2] use (wet + dry) inorganic N (NH_4^+-N plus NO_3^--N) deposition as annual inputs and stream inorganic N exports as annual outputs. We assumed that both watersheds received similar amounts of ambient N deposition, with West Bear receiving an additional 25.2 kg N ha^{-1} yr^{-1} as treatment. These calculations do not account for forest canopy influence on inputs or fluxes such as N fixation or denitrification, both of which are likely small. These unaccounted fluxes are discussed in detail in the Discussion section.

Table 1. Estimated input and output fluxes of inorganic N for East Bear and West Bear watersheds. Input–output fluxes are calculated for the 5-yr period from 2010 to 2014.

Input–output fluxes	East Bear Brook	West Bear Brook
	kg N ha^{-1} yr^{-1}	
Input (deposition)	2.0	27.0
Output (stream export)	0.2	7.0
Retention (input–output)	1.8	20.0

Ecosystem Nitrogen Pools

Samples for chemical characterization of soils, trees, loose litter, and ground vegetation were collected from 2010 to 2014. We calculated ecosystem N pools for each component by multiplying the mass per area (kg ha^{-1}) by total N concentrations (%).

Soils

Soils were collected from 40 quantitative pits (ten in each of four compartments) in the summer of 2010 using methods comparable to those of SanClements et al. (2010). The pits were excavated to the bottom of the solum (1 m) or to contact with basal till or bedrock. Organic soils (O horizon) were quantitatively removed to the top of the mineral soil within 71 cm by 71 cm frames. Mineral soil (B horizon) was quantitatively removed in depth increments 0 to 5 cm, 5 to 25 cm, and 25 cm to the top of the C horizon. Grab samples were collected from the C horizon. Where present, the E horizon depth was measured but not sampled for chemistry. Soils were transported to the laboratory, air-dried, sieved (<6 mm screen for organic soil, and <2 mm screen for mineral soil), and analyzed for total N concentrations using a LECO CN-2000 Analyzer at the Maine Agriculture and Forest Experiment Station (MAFES) Analytical Laboratory. The soil total N concentrations are presented in Supplemental Table S4. The soil N pool was calculated as the sum of organic (coarse and fine) and mineral (fine) fraction.

Tree Biomass

Tree biomass for leaf, branch, stem, and stump-root components was calculated using allometric equations (Young et al., 1980):

$$\ln(\text{biomass, lbs}) = B_0 + B_1 * \ln(\text{DBH, in}) \quad [3]$$

where DBH is the diameter at breast height, and B_0 and B_1 are species-specific coefficients for each tree component (Supplemental Table S1). Tree DBH was measured in July 2014 for 861 trees in 20 plots of 400 m^2, centered around and inclusive of the five cluster plots in each compartment. Biomass was converted to kilograms for further calculation and analysis. The stem biomass values were divided into wood and bark using data from Freedman et al. (1982), who reported that wood comprised 88% of total stem biomass in maple–beech–birch stands, and 87% of total stem biomass in red spruce–balsam fir stands. Individual tree biomass values within each 400 m^2 plot were added to determine biomass per area (kg 400 m^{-2}) and then scaled up to hectare (kg ha^{-1}). These values (kg ha^{-1}) were averaged to calculate compartment-level estimates. Watershed-level (East Bear and West Bear) biomass was calculated as weighted averages of the compartment-level values, using the relative distribution of the two forest types in each watershed (Wang and Fernandez, 1999). The forest cover in West Bear was 51.6% hardwood and 48.4% softwood, and in East Bear was 63.7% hardwood and 36.3% softwood.

Tree Chemistry

Foliage was sampled from the mid-crown in the upper canopy (i.e., sun-lit leaves) using pole pruners in August of 2010, 2012, and 2013. Five trees of each dominant species (American beech, sugar maple, red maple, yellow birch, and red spruce) were sampled in each watershed (i.e., 25 trees per watershed) each year. Leaves showing signs of disease were excluded. Foliar samples were rinsed with deionized water and blotted dry. Wood and bark samples were obtained from tree cores collected in 2011. One hundred trees were sampled across the five dominant species in each watershed. For a subset of these cores, rings from 2009 to 2011 were processed for wood N. Stump–root and branch samples were not collected for chemical analysis. We used wood N values for stump–roots and woody litterfall (i.e., twigs) values for branches. Woody litter was collected during 2012 and 2013, using four traps (52.5 cm by 37 cm) in each cluster plot. The samples were dried and sorted into hardwood and softwood tissues prior to analysis.

Loose Litter

Loose litter was surface litter tissues that were not yet mechanically attached to the forest floor and could be readily brushed aside without resistance. Loose litter was collected during the growing season of 2012. Samples were collected in triplicate from each cluster plot using a 10 cm by 10 cm template, and pooled for each cluster plot for chemical analysis.

Understory Vegetation

Understory vegetation was sampled during the growing season of 2012. The five most dominant species (<60 cm high) were clipped at the ground surface from a 5 m by 5 m area within each cluster plot. The samples were rinsed with deionized water and blotted dry separately by species.

All vegetation samples were dried in a drying room at ~60°C, weighed, ground to <1 mm using a Wiley mill, ground in a ball mill, and analyzed for total N using an elemental analyzer at the University of California Davis Stable Isotope Facility. The N concentrations for each vegetation component by species are reported in Supplemental Table S3.

Tree Nitrogen Accumulation

The tree increment cores collected (from 100 trees) in 2011 were used to determine increment growth. Increment cores were digitized and ring widths were measured using WinDENDRO (Regent Instruments, Inc.). The average ring width values for 2007 to 2011 were used as estimates of recent tree ring increment for the five dominant tree species. Tree ring increment data were used to "hindcast" historical annual biomass accumulation based on the 2014 values (Supplemental Table S2). To calculate N accumulation rates in trees, we multiplied the biomass growth rate by the average N concentrations described above. These N concentration values represent the 2014 sampling effort, and N accumulation rates in trees are therefore a result of biomass accumulation.

Statistical Analysis

Two-tailed *t* tests were used to compare total N concentrations and accumulation rates between watersheds for each given ecosystem component (Tables 2 and 3). Statistical significance was determined at $\alpha = 0.05$. Linear regressions were performed to compare annual stream N exports with temperature and precipitation, and we report the Pearson correlation coefficients (r) for these relationships, with r values >0.50 indicating strong correlation and $p < 0.05$ indicating statistical significance for strong correlations. All statistical analyses were conducted using JMP software, Version 13.

RESULTS

Nitrogen Deposition

Total ambient N deposition (wet + dry) in East Bear and West Bear, as NH_4^+ plus NO_3^- deposition, ranged from 2.3 to 5.4 kg N ha^{-1} yr^{-1} (mean 3.7 kg N ha^{-1} yr^{-1}) for 1990 to 2014 (Fig. 2 and 3). Calculated dry deposition accounted for ~2 to 5% of the annual ambient N inputs during this period. Total ambient NH_4^+ deposition (wet + dry) in East Bear ranged from 1 to 2 kg N ha^{-1} yr^{-1} and was 28 to 49% of the total annual inorganic N deposition. Total inorganic N inputs (ambient + treatment) in West Bear ranged from 27.5 to 30.9 kg N ha^{-1} yr^{-1} (mean 29.0 kg N ha^{-1} yr^{-1}) and NH_4^+ accounted for 88 to 95% of total inorganic N inputs. Total ambient inorganic N deposition declined significantly over the study period from 1990 to 2014 ($r = -0.6291$, $p = 0.0006$) attributable to a 60% decline in NO_3^- deposition ($r = -0.7587$, $p < 0.0001$), while NH_4^+ deposition remained unchanged ($r = -0.2253$, $p = 0.2684$).

Stream DIN Export

Stream DIN exports ranged from below-detection to 2.1 kg N ha^{-1} yr^{-1} (mean 0.4 kg N ha^{-1} yr^{-1}) in East Bear and from 2 to 10 kg N ha^{-1} yr^{-1} (mean 5.6 kg N ha^{-1} yr^{-1}) in West Bear (Fig. 2 and 3). Exports from both watersheds were dominated by NO_3^- (~86 and ~98% of annual inorganic N export in East Bear and West Bear, respectively). In East Bear, consistent

Table 2. Nitrogen pools in ecosystem components in each compartment of N treatment and forest type (East Bear hardwoods, East Bear softwoods, West Bear hardwoods, and West Bear softwoods) after 25 yr of N additions.

	East Bear Brook (reference)		West Bear Brook (treated)	
Pool	Mean	SE	Mean	SE
	kg N ha^{-1}			
		Hardwoods		
Total vegetation N	640	53	969*†	69
Foliage	100	7	141*	15
Woody biomass	459	38	710	66
Branches	103	8	207*	25
Wood	207	19	261	25
Bark	78	6	151*	18
Stump-root	71	6	90	9
Ground vegetation	0.19	0	0.33	0
Loose litter	81	11	118	15
Total pedon N	8170	1325	8157	753
Organic soil	1781	228	1729	326
Mineral soil	6389	1215	6428	816
		Softwoods		
Total vegetation N	1028	59	1311*	78
Foliage	269	18	347*	22
Woody biomass	607	51	837	52
Branches	189	15	215	14
Wood	248	21	381*	24
Bark	71	7	88	5
Stump-root	99	8	152*	10
Ground vegetation	0.24	0	2.20	2
Loose litter	153	33	126	27
Total pedon N	10018	1707	10421	910
Organic soil	2422	335	3523*	364
Mineral soil	7597	1625	6898	1020
		Total N in watershed (hardwood and softwood, vegetation and soil)‡		
	9623		10387	

† Two-tailed *t* tests were used to examine differences between watersheds; asterisks denote significant differences between watersheds at $\alpha = 0.05$.
‡ Total N in the watershed refers to the sum of soil and biomass N pools, as aerially weighted averages of the hardwoods and softwoods.

Table 3. Estimated biomass increment N accumulation for East Bear and West Bear watersheds. Tree biomass N accumulation is calculated using tree increment cores for the 5-yr period from 2007 to 2011.

	East Bear Brook (reference)		West Bear Brook (treated)	
Tree biomass N pool	Mean	SE	Mean	SE
	kg N ha^{-1} yr^{-1}			
		Hardwoods		
Total tree biomass	15.5	1.2	22.3*†	1.2
Leaves	2.6	0.2	3.3*	0.2
Branches	3.0	0.4	5.7*	0.4
Wood	5.7	0.30	6.8*	0.3
Bark	2.3	0.3	4.2*	0.3
Stump–root	1.9	0.1	2.3*	0.1
		Softwoods		
Total tree biomass	24.6	1.9	24.1	1.9
Leaves	7.3	0.6	6.7	0.6
Branches	5.4	0.4	4.5	0.4
Wood	7.1	0.6	7.9	0.6
Bark	2.0	0.1	1.9	0.1
Stump–root	2.8	0.2	3.1	0.2
		Entire watershed (hardwoods and softwoods)‡		
	18.8		23.2	

† Two-tailed *t* tests were used to examine differences between watersheds; asterisks denote significant differences between watersheds at $\alpha = 0.05$.
‡ Biomass accumulation rates for the entire watershed were calculated as aerially weighted averages of the hardwoods and softwoods.

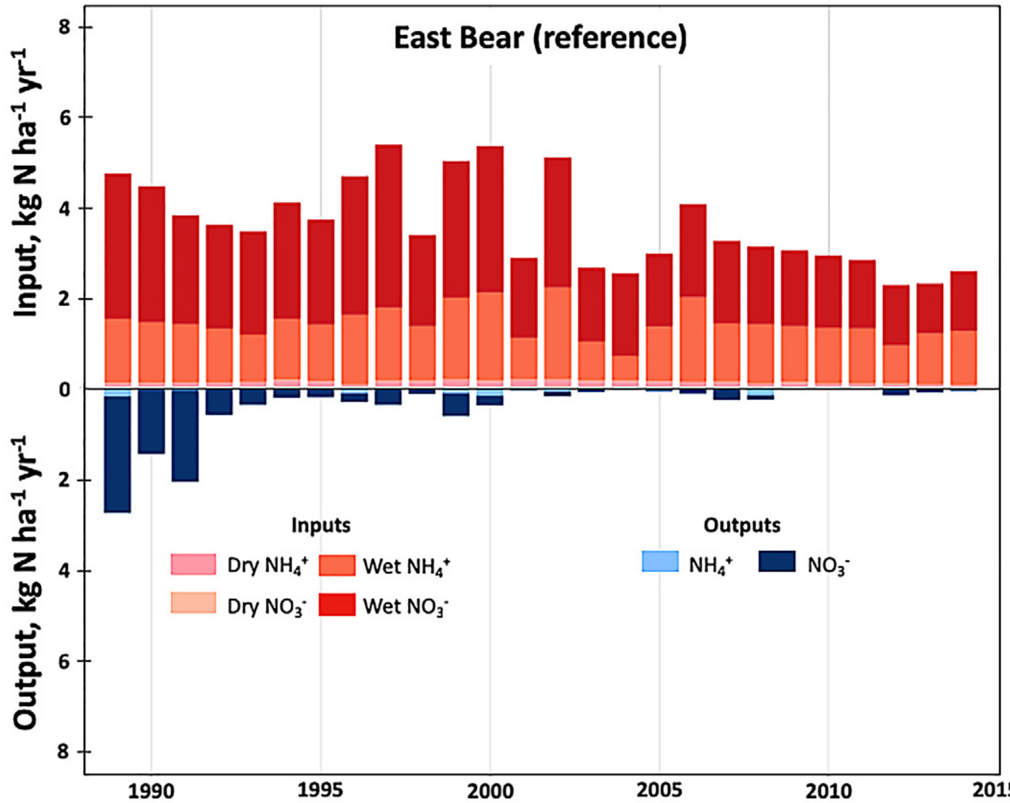

Fig. 2. Input and output inorganic N fluxes for East Bear Brook (reference watershed). The *x* axis represents the water year, October 1 to September 30.

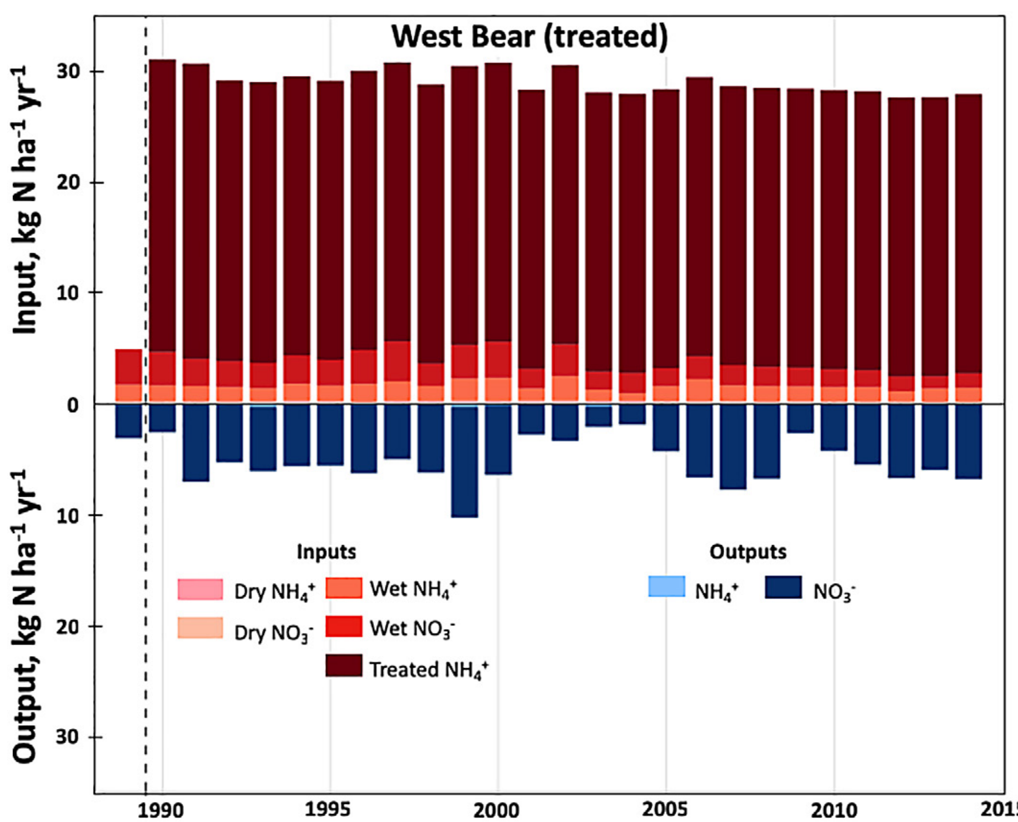

Fig. 3. Input and output inorganic N fluxes for West Bear Brook (treated watershed). The *x* axis represents the water year, October 1 to September 30. The dashed vertical line represents onset of treatment in West Bear Brook.

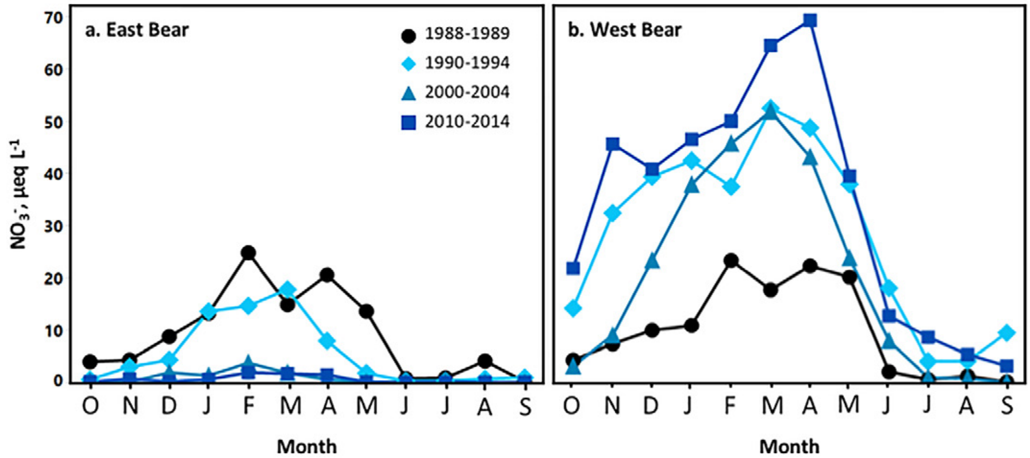

Fig. 4. Monthly stream NO_3^- concentrations during the study period for (a) East Bear Brook (reference watershed) and (b) West Bear Brook (treated watershed). 1988 to 1989 represents pre-treatment data for West Bear. Data for 1990 to 2014 represent 5-yr intervals during the first, second, and third decades of manipulation.

with temporal trends in ambient N deposition, inorganic N exports declined significantly (87% decline) from 1990 to 2014 ($r = -0.6557$, $p = 0.0005$), but there was no statistically significant long-term trend in West Bear ($r = 0.0239$, $p = 0.9119$). Over the course of the study, East Bear retained 83 to nearly 100% (mean 94% ± SE 2.5) or ~3 kg N ha^{-1} yr^{-1}, and West Bear retained 66 to 93% (mean 81% ± SE 1.4), or ~23 kg N ha^{-1} yr^{-1}, of total inorganic N inputs. Table 1 shows recent (2010–2014) input-output fluxes and mass balance of inorganic N for both watersheds. Stream NO_3^- concentrations showed a distinct seasonal pattern (Fig. 4), peaking during the spring months (2.21 µeq L^{-1} ± SE 0.24 in East Bear and 62.7 µeq L^{-1} ± SE 19.8 in West Bear, for 2010–2014) and decreasing to 'base flow' concentrations in the summer months (below-detection in East Bear and 8.83 µeq L^{-1} ± SE 1.94 in West Bear, for 2010–2014).

Ecosystem Nitrogen Pools

Figure 5 shows pools of N for each watershed, and Table 2 shows N pools by compartment 20 to 25 yr after the fertilizer treatment was initiated. The vegetation N pool accounted for ~7 to 11% of total ecosystem N, across both forest types and both watersheds (Table 2). Overall, vegetation in West Bear had ~1.3-fold more N per hectare than in East Bear. Tree foliage accounted for ~15% of the total vegetation N pool in the deciduous forests, and ~26% in the coniferous forests. This was consistent between the reference and the treated watersheds (Table 2). Both foliage and woody biomass had ~1.4-fold more N in West Bear than in East Bear, but the ground vegetation and leaf litter N pools did not differ between the two watersheds. Trees in hardwood stands had greater N accumulation rates in West Bear than in East Bear (Table 3), whereas this was not the case in softwood stands. Overall, biomass accumulation of N was ~19 kg N ha^{-1} yr^{-1} in East Bear and ~23 kg N ha^{-1} yr^{-1} in West Bear for the period 2007 to 2011.

The largest pool of N among the ecosystem components was the mineral soil (>6600 kg N ha^{-1} in both watersheds, 63–70% of total ecosystem N), although N content did not dif-

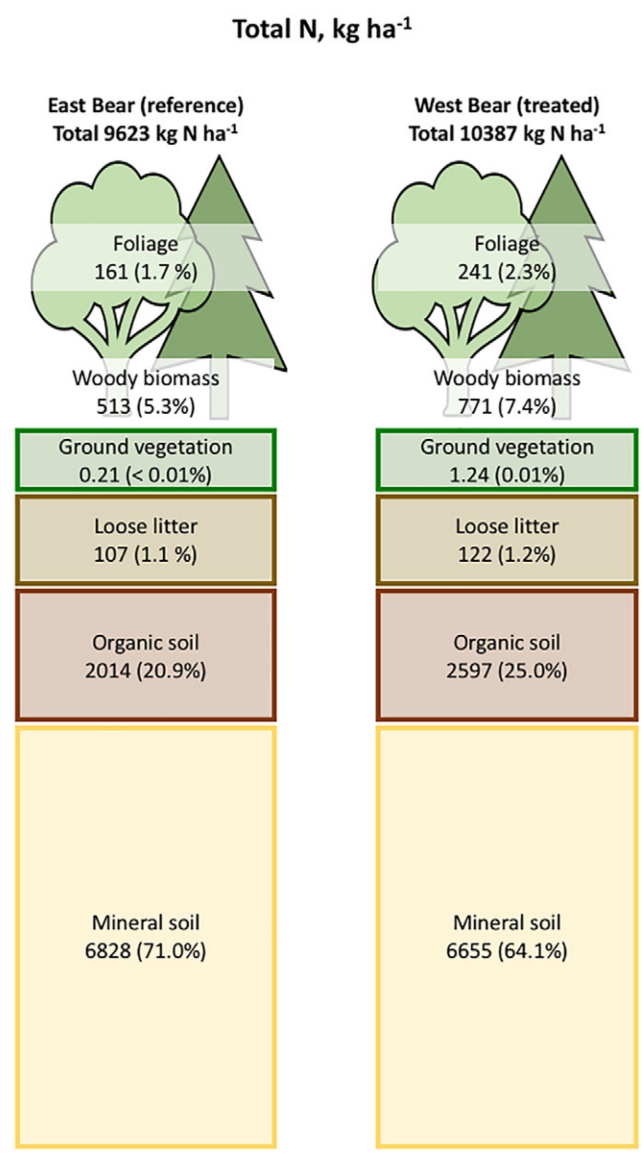

Fig. 5. Ecosystem N pools (kg N ha^{-1}) at Bear Brook Watershed in Maine after 25 yr of N additions. Values in parentheses reflect percentages of the total watershed N pool.

fer significantly between the two watersheds (Table 2; Fig. 5). In the softwood stands, organic horizons in West Bear had ~1.5-fold more N compared to East Bear, but in the hardwood stands, organic horizon N content did not differ significantly between watersheds (Table 2).

DISCUSSION

Watershed Dissolved Inorganic N Retention

The high DIN retention in the reference East Bear was likely driven by low N deposition and a high C to N ratio in these soils. Nitrogen is generally retained in forest systems via microbial immobilization (Vitousek and Matson, 1984; Johnson et al., 2000; Perakis et al., 2005; Templer et al., 2005; Piatek and Adams, 2011), incorporation into plant biomass (Fenn et al., 1998; Templer et al., 2005; Emmett, 2007; Schleppi et al., 2017), or abiotic immobilization, mainly incorporation into soil organic matter (Johnson et al., 2000; Fitzhugh et al., 2003; Perakis et al., 2005). The higher N exports in West Bear likely reflected that these N retention mechanisms were exceeded by the higher N inputs. Additions of NH_4^+ stimulate nitrification by providing labile NH_4^+ substrate, and also inhibit plant NO_3^- uptake by suppressing the production and activity of NO_3^- assimilatory enzymes (Emmett, 2007) and by suppressing the N-uptake activity of mycorrhizae (Högberg et al., 2007). These mechanisms accelerate NO_3^- leaching losses. At the NITREX study sites across Europe, Dise and Wright (1995) found that NO_3^- leaching increased with increasing N deposition, showing that the N retention mechanisms were stronger at sites receiving <10 kg N ha^{-1} annually. At the Fernow Experimental Forest in West Virginia, USA, Gilliam et al. (2018) found that although stream NO_3^- increased, soil nitrification rates did not consistently increase in response to N additions. Instead, they attributed increased stream NO_3^- exports to reduced biological uptake. At our site, we observed greater nitrification rates in West Bear than in East Bear (Patel and Fernandez, 2018), which likely contributed partially to the elevated stream exports.

Long-term Watershed Input–Output N Fluxes

Significant declines in ambient atmospheric N deposition at BBWM from 1990 to 2014 were attributable to a combination of (i) emissions reductions from stationary sources resulting from air pollution policies enacted during this period, such as Titles IV (SO_2, NOx) and VI (ozone) of the US Clean Air Act Amendments (CAAA) of 1990 and Phase I controls of the CAAA in 1995 (Du et al., 2014; Kahl et al., 2004); and (ii) reduced vehicular emissions in the northeastern United States (Butler et al., 2003).

Stream DIN exports in East Bear declined from 1989 to 1993, corresponding to a regional decline of elevated surface water NO_3^- concentrations, following severe winter frost prior to 1989 (Mitchell et al., 1996). This short-term phenomenon ended in ~1993 as East Bear exports returned to chronically low values. Subsequent declines in East Bear stream NO_3^- largely reflected the long-term trends in declining N inputs, and immobilization in the soil. The experimental N additions (25.2 kg N ha^{-1} yr^{-1}) were originally thought to represent a fourfold increase in N loading in West Bear compared to East Bear, using measured values of wet deposition and estimates of dry deposition at the beginning of the study (Norton et al., 1999). The data reported here suggest ambient plus experimental inputs to West Bear may have been up to ~7-fold the East Bear inputs at the start of the experiment, because of improvements in the estimated dry deposition of N. With declining ambient N deposition, West Bear was receiving up to ~12-fold the East Bear N inputs by 2014 as a result of declining ambient N deposition. Experimental N additions increased NO_3^- exports in West Bear within four months after the start of the experiment. Tracer ^{15}N experiments in 1991 to 1993 revealed that most of the initial excess stream NO_3^- exports did not originate from the added NH_4^+. Rather, the treatments stimulated nitrification of the resident, 'old' N in the ecosystem (Nadelhoffer et al., 1999). Changes in the isotopic signature of stream NO_3^- indicated a greater contribution of the added fertilizer to stream NO_3^- after 1993. This is consistent with N mineralization results from BBWM indicating that N mineralization and nitrification rates also increased in West Bear after 1993 (Fernandez et al., 2000; Jefts et al., 2004; Patel and Fernandez, 2018; Wang and Fernandez, 1999).

Long-term patterns of DIN export in the West Bear stream revealed significant inter-annual variability. Stream N export was not strongly correlated with mean annual precipitation (Pearson's r = 0.36, p = 0.007) or air temperatures (Pearson's r = 0.33, p = 0.010). Therefore, other factors likely contributed to the inter-annual variability evident in the long-term record, some of which were stochastic disturbance events. For example, higher DIN exports occurred in 1999, evident in both watersheds, which we attribute to the ice storm of 1998 that deposited ~13 cm of ice in central Maine, a major disturbance event for Maine forests (Irland, 1998). Some damage to the forest canopy at BBWM as a result of the 1998 ice storm occurred in components of the watershed. Anecdotally, greater damage occurred in the hardwood components, although no quantitative studies were done. The ice storm resulted in increased litter inputs to the forest floor and increased forest floor insolation because of damages to the canopy. In combination, these altered processes resulted in a short pulse of DIN losses in BBWM streams (Houlton et al., 2003; SanClements et al., 2010). Additional disturbances from wind storms and defoliations by caterpillar infestations occurred over the period of record, but these were not quantitatively assessed for their effects on the long-term DIN export record.

Ecosystem Nitrogen Mass Balance

The relative sizes of ecosystem N pools at BBWM reflect the general distribution of N common in temperate upland forests of northeastern North America, with soils comprising the biggest N pools (e.g., Mitchell et al., 1992; Kelly et al., 2011; Yanai et al., 2013). During the 25 yr of treatment, the cumulative retention of DIN in East Bear was 81 kg N ha^{-1} and the cumulative retention of DIN in West Bear was 582 kg N ha^{-1}.

In the reference East Bear watershed, the vegetation N accumulation rate was an order of magnitude greater than the annual ecosystem N retention (Tables 1 and 3), revealing an additional source of N, ~17 kg N ha^{-1} yr^{-1}, beyond atmospheric N contributions. Trees in northeastern North America have been reported to mine N from the mineral soil during periods of stand development, i.e., the soil acts as a net source of N (Hooker and Compton, 2003; Yanai et al., 2013). Mineral soil N is present as plant-available inorganic forms (NH_4 and NO_3), or as organic N forms in microbial biomass and soil organic matter, the latter being by far the larger pool (Jefts et al., 2004). Organic N can be mineralized into plant-available forms in support of plant uptake. Mining of inorganic N from soil was documented at BBWM in the early 1990s (Nadelhoffer et al., 1999), and our calculations suggest a net soil N uptake by vegetation persists to at least 2014.

Tracer ^{15}N isotopic studies during 1991 to1993 showed that the soil was the major sink of the added labeled N (Nadelhoffer et al., 1999). Based on mass balance calculations for the data presented in this paper, soils continued to be a major sink of the added N even after 25 yr of N enrichment. However, the relative size of the total soil N pools (Fig. 5; Table 3) and the high spatial variability in our soils (see the SE values in Table 1) limit our statistical power to observe significant differences between East Bear and West Bear soil N pools. An analysis of temporal trends in soil total N concentrations also showed that West Bear soil N concentrations did not change through time despite 25 yr of N amendments (Patel and Fernandez, 2018). The minimum detectable change (for 95% confidence intervals) for these data was ~380 kg N ha^{-1} for organic horizons, 1175 kg N ha^{-1} for mineral horizons, and 1234 kg N ha^{-1} for the total pedon. This suggests that even if all the N retained in West Bear (582 kg N ha^{-1}) were stored in the soil, we would not be able to statistically detect the change in total soil N from the treatment in West Bear soils based on the sampling intensity in this study.

In the softwood stands, West Bear organic horizons had significantly more total N than East Bear (Table 2), because West Bear soils had a significantly greater soil mass (Supplemental Table S2). Nitrogen enrichment suppresses organic matter decomposition, particularly for high-lignin litter (Edwards et al., 2011; Frey et al., 2014; Morrison et al., 2018; Van Diepen et al., 2015). Slower decomposition of high-lignin softwood litter in West Bear could contribute to the thicker horizons in these stands. This is consistent with findings by Tatariw et al. (2018) that N additions increased C-limitation for microbes in West Bear organic horizons, indicating that the N additions had altered the decomposer community. In contrast, neither organic horizon thickness nor total N differed between watersheds in the hardwood stands, suggesting that if N additions were suppressing rates of organic matter decomposition in softwood stands, hardwood stands did not reflect a similar response. Experimental N additions also increased litterfall mass in softwood stands but not in hardwood stands at the Harvard Forest (Frey et al., 2014). It is possible that increased litter inputs could also have contributed to the thicker organic horizons in the West Bear softwood stands at BBWM.

Our results highlight the importance of forest type on net ecosystem N retention, consistent with previous studies that demonstrated species-specific effects on N cycling (Booth et al., 2005; Oulehle et al., 2018; Templer et al., 2005). The ~45% greater biomass N increment (Table 3) in West Bear hardwood compared to East Bear hardwood stands suggests N additions reduced N limitations resulting in accelerated biomass and N annual increment in the treated watershed. In contrast, biomass N increment was similar between West Bear softwoods and East Bear softwoods (Table 3). It is possible that experimental N additions in West Bear, combined with the relatively low rates of historical atmospheric deposition compared to more highly impacted sites farther southwest (McNulty et al., 2007; Pardo et al., 2011) were not sufficient to increase soil N availability to a level that resulted in a detectable increase in softwood growth. Another possible reason for the lack of softwood response is limitation by other nutrients. Elvir et al. (2010) attributed the apparent lack of tree growth response to added N at BBWM to the lower base cation availability in the West Bear softwood soils, caused by the N and S additions. Indeed, studies previously conducted at BBWM provide evidence that softwoods had approximately half the soil exchangeable Ca and Mg of hardwood soils (Fernandez et al., 2003; SanClements et al., 2010), suggesting other limiting nutrients in softwood stands may be limiting the growth response to N additions. While the focus of this study is on the effects of long-term N additions at BBWM, the treatment also added S which played an important role in altering conditions in West Bear such as soil base cations and Al. The effects of the S additions on the responses in ecosystem N dynamics reported here are poorly understood and are not explored in this analysis.

Despite no significant increase in biomass N accumulation in the softwoods (Table 3), the softwood stands in West Bear did show significantly greater biomass N pools in 2014 (Table 2). This suggests that there may have been a growth response in the softwood stands in the earlier years of the experiment that are no longer occurring, although we do not have empirical data to quantify this pattern over time. The lack of growth response in the softwood stands suggests that if the added N was not taken up by the trees, it was either retained in the soil or lost via leaching. Studies conducted using ion-exchange resins in soils (Szillery et al., 2006) and lysimeters (Fatemi et al., 2012) demonstrated that NO_3^- leaching was greater in West Bear softwoods than in West Bear hardwoods reflecting the consequence of the lower N uptake in softwoods and the relative absence of ground vegetation compared with the hardwood stands.

Other studies reporting increased foliar and/or total biomass N accumulation in conifers [e.g., Harvard Forest, USA (Magill et al., 2000) and Gårdsjön, Sweden (Kjønaas and Stuanes, 2008)] added significantly more N to their ecosystems (≥50 kg N ha^{-1} yr^{-1}, compared to 25 kg N ha^{-1} yr^{-1} at BBWM). In contrast to these studies, Lovett et al. (2013) found that additions of 50 kg N ha^{-1} yr^{-1} did not alter aboveground biomass growth in the Catskill Mountains of New York, USA, likely because of the high historical N deposition, the advanced age of the forest, and/or limitation by some other nutrient.

Nitrogen Saturation and Critical Loads Exceedance

Substantial research has been conducted over more than three decades to improve our understanding of forest ecosystem effects of N enrichment, including the development of the concept of N saturation, and the identification of important thresholds of ecological and biogeochemical responses. The conceptual model of Aber et al. (1989, 1998) described N saturation in terms of N mineralization and plant productivity. According to this framework, the reference East Bear watershed was in Stage 0 (N limitation of plant growth and decomposition), and the treated West Bear was in Stage 1 (increased foliar N concentrations and increased biomass) during the third decade of treatment. The conceptual model of Stoddard (1994) defined N saturation in terms of ecosystem N retention and NO_3^- export. At the start of this study (1988–1999), both East Bear and West Bear were in Stage 1 (higher spring NO_3^- concentrations compared to stage 0), reflecting the transient winter frost effects discussed earlier during the initial years of this study. Subsequent declines in East Bear stream NO_3^- resulted in East Bear transitioning back to a Stage 0 status (low NO_3^- concentrations during base flow, with higher concentrations during snowmelt and spring runoff) by the mid-1990s, which persisted to at least 2014. In West Bear, N additions increased NO_3^- exports almost immediately, and the watershed has remained at Stage 2 (increased baseflow NO_3^- concentrations) through 2014. Despite chronic multi-decadal elevated N inputs to West Bear, the watershed has not progressed over time to more advanced stages of N enrichment, and has not shown increases in tree mortality or increasing stream NO_3^- export.

Critical loads have been used to describe thresholds of N deposition above which detrimental ecological effects are reported (Pardo et al., 2011). Critical loads of 8 to 10 kg N ha^{-1} yr^{-1} are generally recognized for significant NO_3^- leaching in temperate forests of North America and Europe (Campbell et al., 2004; Dise et al., 2009; Pardo et al., 2011; Watmough et al., 2005). Our treatment of 25.2 kg N ha^{-1} yr^{-1} was more than double this threshold and, as expected, we saw increased stream NO_3^- exports in West Bear (Fig. 3 and 4). The low critical loads for mycorrhizae and lichens (<7 kg N ha^{-1} yr^{-1}; Pardo et al., 2011) suggest that the community structure and functions of these indicators have likely been altered in West Bear. Studies are currently underway to examine mycorrhizal communities at BBWM.

Over the past three decades, improvements in air quality have reduced N deposition in the northeastern United States, potentially altering the N status of temperate forest ecosystems (Groffman et al., 2018; Patel and Fernandez, 2018). The concomitant increases in air temperatures and atmospheric carbon dioxide concentrations (Wuebbles et al., 2017) have stimulated biomass growth (Kosiba et al., 2018), potentially increasing plant N demand. There is also evidence of increased C flow to soils increasing soil C availability and potentially stimulating microbial N immobilization in temperate forests (Groffman et al., 2018). The increased biological N uptake, combined with reduced atmospheric N inputs, have reduced the available N in soils, and there is accumulating evidence for the oligotrophication of these temperate forests (McLauchlan et al., 2017; Craine et al., 2018; Groffman et al., 2018). The results reported here also suggest that these forests may be transitioning to greater N limitation over time, and long-term studies like BBWM are essential to understand how these changes are altering forest productivity and forest health.

Missing Nitrogen Fluxes

Our calculations of net ecosystem N retention in East Bear (94% ± SE 2.5) and West Bear (81% ± SE 1.4) used the input–output approach commonly seen in the literature that relies on inorganic N inputs in atmospheric deposition and exports via surface waters in recently unharvested ecosystems. A number of components of the N cycle are commonly not included in these analyses because the data are unavailable and measurements are often difficult to quantify. For the BBWM N mass balance, N inputs represent above-canopy atmospheric inputs of inorganic N and do not reflect the interaction of the canopy with deposition processes (Decina et al., 2018; Fernandez et al., 1999; Lovett and Lindberg, 1993; Weathers et al., 2006). Throughfall measurements at BBWM and elsewhere in Maine have highlighted the significant contribution of fog deposition and forest canopy interception to ecosystem N inputs (Fernandez et al., 1999; Norton et al., 1999; Weathers et al., 2006). At Acadia National Park in coastal Maine, total deposition inputs of N were found to be ~4-fold the bulk precipitation inputs (Weathers et al., 2006). If a similar "scaling factor" of 4 were to hold true for BBWM, it would suggest an important additional N input to the budget (~6 to 8 kg ha^{-1}). Biological N fixation can also contribute small additional inputs of atmospheric N in these temperate forest ecosystems. Studies in northern hardwood forests at the Hubbard Brook Experimental Forest in nearby New Hampshire reported ~0.5 kg N ha^{-1} yr^{-1} fixed in 40- to 60-yr old stands (Roskoski, 1980). Recent studies have reported that rock weathering may contribute as much as 15% to ecosystem N inputs (Houlton et al., 2018) in temperate ecosystems. However, this is likely not a significant source of N at BBWM because of the low N concentration of the bedrock and till rock components (<0.1% by weight, unpublished data), and low weathering rates of the till and bedrock at BBWM (Lawrence et al., 1997). We did not include dissolved organic N (DON) inputs in our calculations, but DON likely did not contribute significantly to the N inputs at BBWM. At Hubbard Brook, DON was ~10% of bulk N precipitation (Yanai et al., 2013), and limited collections at BBWM in the 1990s showed that DON was below-detection in precipitation at BBWM (unpublished data).

Similarly, our estimates of total N export do not include measurements of DON and denitrification. Limited measurements of DON conducted in East Bear and West Bear streams during the study period (data not shown) indicated that DON concentrations were similar in both streams, approximately double the inorganic N concentrations in East Bear. But the limited period of DON measurements precludes our quantitative estimate of DON export over the full time series presented in this study. Denitrification fluxes were measured at BBWM during 2000 to 2001 (Venterea et al., 2004) and 2010 to 2011 (Morse et al., 2015). Venterea et al. (2004) found that nitric oxide fluxes were negligible in East Bear and <0.4 kg N ha^{-1} yr^{-1} in West Bear (~1.6% of the annual inputs), while Morse et al. (2015) found that (N_2 + N_2O) flux was <1 kg N ha^{-1} yr^{-1} in East Bear and West Bear and did not differ between watersheds. Both studies found that denitrification was a relatively small export of N from our ecosystem. Isotopic analyses of groundwater at Hubbard Brook suggest that traditional estimates of denitrification may underestimate total denitrification fluxes from these northern temperate forested ecosystems (Wexler et al., 2014).

This discussion of missing N fluxes in this study suggests that missing N inputs, driven primarily by a deposition scaling factor, are larger than the missing exports, and therefore the realized N retention in these ecosystems is actually greater than the calculated retention reported here.

Implications for Nitrogen Cycling

Our understanding of ecosystem processes at BBWM assumes that East Bear and West Bear were similar in terms of soil (mass and chemistry), vegetation (biomass and chemistry), and stream (discharge and nutrient export) characteristics prior to treatment. The streams were the only ecosystem component monitored prior to the beginning of the West Bear manipulation, and the site was chosen because of the comparable hydrology and chemistry of the streams, including N export, and general similarity in aspect, topography, watershed size, soils, and vegetation (Fig. 4; Norton et al., 2010). We added 25.2 kg N ha^{-1} yr^{-1} to West Bear, significantly more than the mean ambient deposition of 3.7 kg N ha^{-1} yr^{-1} during the 25-yr study period, making this chemical perturbation the likely primary factor driving ecosystem responses reported here, rather than pre-treatment differences between the watersheds. The ecosystem pools (Fig. 5) indicate that West Bear had ~700 kg N ha^{-1} more than East Bear, reasonably similar to the net excess retention of ~20 kg N ha^{-1} yr^{-1} over 25 yr. While it is unlikely that these rates of accumulation were consistent over this time period, the agreement of these estimates lends support for the dominant role of the experimental N additions being responsible for altered patterns of N cycling in West Bear.

The contrasting responses between hardwood and softwood stands at BBWM highlight the critical role of forest composition in determining the response of forests to elevated N inputs. The lack of a clear biomass N accumulation response and the higher N leaching response in softwood stands, compared to hardwoods, suggests a greater likelihood for associated surface water impacts in softwood dominated landscapes. Twenty-five years of chronic whole watershed N additions had a positive effect on forest growth in hardwoods, a negative consequence in stream water chemistry, and a relatively rapid emergence of these trends early in the experiment that have persisted in a dynamic equilibrium for 25 yr. We suggest this dynamic equilibrium will persist until stochastic disturbance events (e.g., ice storms, wind, insect and disease, fire) or the emergence of a later successional stage reduces the aboveground biomass uptake of soil N.

ACKNOWLEDGMENTS

This study was supported by grants from the National Science Foundation (DEB-1119709 and DEB-1704200) and the Maine Agricultural and Forest Experiment Station (MAFES #ME041507). We thank Cheryl Spencer, Mike Handley, and numerous undergraduate and graduate students for their assistance in the laboratory and field, and with data handling. This is MAFES publication 3657.

SUPPLEMENTAL MATERIAL

Supplemental material is available with the online version of this article. The supplemental document contains four tables: Table S1, Coefficients for allometric equations; Table S2, Tree biomass and biomass accumulation values, and oven-dried soil mass values for the two watersheds; Table S3, Tissue N concentration in vegetation at BBWM (measured for the period 2010–2013); and Table S4, Total N concentration by soil increment for each compartment at BBWM (for the 2010 quantitative pits).

REFERENCES

Aber, J.D., W. Mcdowell, K.J. Nadelhoffer, A.H. Magill, G. Berntson, S.G. McNulty, et al. 1998. Nitrogen saturation in temperate forest ecosystems: Hypotheses revisited. Bioscience 48:921–934. doi:10.2307/1313296

Aber, J.D., K.J. Nadelhoffer, P. Steudler, and J.M. Melillo. 1989. Nitrogen saturation in northern forest ecosystems. Bioscience 39:378–386. doi:10.2307/1311067

Booth, M.S., J.M. Stark, and E. Rastetter. 2005. Controls on nitrogen cycling in terrestrial ecosystems: A synthetic analysis of literature data. Ecol. Monogr. 75:139–157. doi:10.1890/04-0988

Butler, T.J., G.E. Likens, F.M. Vermeylen, and B.J.B. Stunder. 2003. The relation between NOx emissions and precipitation NO_3^- in the eastern USA. Atmos. Environ. 37:2093–2104. doi:10.1016/S1352-2310(03)00103-1

Campbell, J.L., J.W. Hornbeck, M.J. Mitchell, M.B. Adams, M.S. Castro, C.T. Driscoll, et al. 2004. Input-output budgets of inorganic nitrogen for 24 forest watersheds in the Northeastern United States: A review. Water Air Soil Pollut. 151:373–396. doi:10.1023/B:WATE.0000009908.94219.04

Carrara, J.E., C.A. Walter, J.S. Hawkins, W.T. Peterjohn, C. Averill, and E.R. Brzostek. 2018. Interactions among plants, bacteria, and fungi reduce extracellular enzyme activities under long-term N fertilization. Glob. Change Biol. doi:10.1111/gcb.14081

Casson, N.J., C.M. Eimers, and J.M. Buttle. 2010. The contribution of rain-on-snow events to nitrate export in the forested landscape of south-central Ontario, Canada. Hydrol. Processes 24:1985–1993. doi:10.1002/hyp.7692

CASTNET. 2018. Dry deposition—Weekly. www.epa.gov/castnet. Clean Air Status and Trends Network. (accessed 1 May 2018).

Craine, J.M., A.J. Elmore, L. Wang, J. Aranibar, M. Bauters, P. Boeckx, et al. 2018. Isotopic evidence for oligotrophication of terrestrial ecosystems. Nat. Ecol. Evol. 2:1735–1744. doi:10.1038/s41559-018-0694-0

Decina, S.M., P.H. Templer, and L.R. Hutyra. 2018. Atmospheric inputs of nitrogen, carbon, and phosphorus across an urban area: Unaccounted fluxes and canopy influences. Earth's Future. 6:134–148. doi:10.1002/2017EF000653

Dise, N.B., J.J. Rothwell, V. Gauci, C. van der Salm, and W. de Vries. 2009. Predicting dissolved inorganic nitrogen leaching in European forests using two independent databases. Sci. Total Environ. 407:1798–1808. doi:10.1016/j.scitotenv.2008.11.003

Dise, N.B., and R.F. Wright. 1995. Nitrogen leaching from European forests in relation to nitrogen deposition. For. Ecol. Manage. 71:153–161. doi:10.1016/0378-1127(94)06092-W

Du, E., W. De Vries, J.N. Galloway, X. Hu, and J. Fang. 2014. Changes in wet nitrogen deposition in the United States between 1985 and 2012. Environ. Res. Lett. 9. doi:10.1088/1748-9326/9/9/095004

Edwards, I.P., D.R. Zak, H. Kellner, S.D. Eisenlord, and K.S. Pregitzer. 2011. Simulated atmospheric N deposition alters fungal community composition and suppresses ligninolytic gene expression in a Northern Hardwood forest. PLoS One 6:1–10. doi:10.1371/journal.pone.0020421

Elvir, J.A., L.E. Rustad, G.B. Wiersma, I.J. Fernandez, A.S. White, and G.J. White. 2005. Eleven-year response of foliar chemistry to chronic nitrogen and sulfur additions at the Bear Brook Watershed in Maine. Can. J. For. Res. 35:1402–1410. doi:10.1139/x05-072

Elvir, J.A., G.B. Wiersma, S. Bethers, and P. Kenlan. 2010. Effects of chronic ammonium sulfate treatment on the forest at the Bear Brook Watershed in Maine. Environ. Monit. Assess. 171:129–147. doi:10.1007/s10661-010-1523-3

Elvir, J.A., G.B. Wiersma, M.E. Day, M.S. Greenwood, and I.J. Fernandez. 2006. Effects of enhanced nitrogen deposition on foliar chemistry and physiological processes of forest trees at the Bear Brook Watershed in Maine. For. Ecol. Manage. 221:207–214. doi:10.1016/j.foreco.2005.09.022

Emmett, B.A. 2007. Nitrogen saturation of terrestrial ecosystems: Some recent findings and their implications for our conceptual framework. Water Air Soil Pollut. Focus 7:99–109. doi:10.1007/s11267-006-9103-9

Fatemi, F.R., I.J. Fernandez, K.S. Simon, and D.B. Dail. 2016. Nitrogen and phosphorus regulation of soil enzyme activities in acid forest soils. Soil Biol. Biochem. 98:171–179. doi:10.1016/j.soilbio.2016.02.017

Fatemi, F.R., I.J. Fernandez, J.E. Szillery, S.A. Norton, and L.E. Rustad. 2012. Soil solution chemical response to two decades of experimental acidification at the Bear Brook Watershed in Maine. Water Air Soil Pollut. 223:6171–6186. doi:10.1007/s11270-012-1348-5

Fenn, M.E., M.A. Poth, J.D. Aber, J.S. Baron, B.T. Bormann, D.W. Johnson, et al. 1998. Nitrogen excess in North American ecosystems : Predisposing factors, ecosystem responses, and management strategies. Ecol. Appl. 8:706–733. doi:10.1890/1051-0761(1998)008[0706:NEINAE]2.0.CO;2

Fernandez, I.J., M.B. Adams, M.D. SanClements, and S.A. Norton. 2010. Comparing decadal responses of whole-watershed manipulations at the Bear Brook and Fernow experiments. Environ. Monit. Assess. 171:149–161. doi:10.1007/s10661-010-1524-2

Fernandez, I.J., L.E. Rustad, M. David, K.J. Nadelhoffer, and M.J. Mitchell. 1999. Mineral soil and solution responses to experimental N and S enrichment at the Bear Brook Watershed in Maine (BBWM). Environ. Monit. Assess. doi:10.1023/A:1006110910473

Fernandez, I.J., L.E. Rustad, S.A. Norton, J.S. Kahl, and B.J. Cosby. 2003. Experimental acidification causes soil base-cation depletion at the Bear Brook Watershed in Maine. Soil Sci. Soc. Am. J. 67:1909–1919. doi:10.2136/sssaj2003.1909

Fernandez, I.J., J.A. Simmons, and R.D. Briggs. 2000. Indices of forest floor nitrogen status along a climate gradient in Maine, USA. For. Ecol. Manage. 134:177–187. doi:10.1016/S0378-1127(99)00256-X

Ferretti, M., M. Calderisi, A. Marchetto, P. Waldner, A. Thimonier, M. Jonard, et al. 2015. Variables related to nitrogen deposition improve defoliation models for European forests. Ann. For. Sci. 72:897–906. doi:10.1007/s13595-014-0445-6

Fitzhugh, R.D., G.M. Lovett, and R.T. Venterea. 2003. Biotic and abiotic immobilization of ammonium, nitrite, and nitrate in soils developed under different tree species in the Catskill Mountains, New York, USA. Glob. Change Biol. 9:1591–1601. doi:10.1046/j.1365-2486.2003.00694.x

Freedman, B., P.N. Duinker, H. Barclay, R. Morash, and U. Prager. 1982. Forest biomass and nutrient studies in central Nova Scotia. Information Report. Maritimes Forest Research Centre, Fredericton, New Brunswick, Canada.

Frey, S.D., S.V. Ollinger, K.J. Nadelhoffer, R. Bowden, E. Brzostek, A. Burton, et al. 2014. Chronic nitrogen additions suppress decomposition and sequester soil carbon in temperate forests. Biogeochemistry 121:305–316. doi:10.1007/s10533-014-0004-0

Gilliam, F.S., C.A. Walter, M.B. Adams, and W.T. Peterjohn. 2018. Nitrogen dynamics in the mineral soil of a central Appalachian hardwood forest during a quarter century of whole-watershed N additions. Ecosystems 21:1–16. doi:10.1007/s10021-018-0234-4

Groffman, P.M., C.T. Driscoll, J. Durán, J.L. Campbell, L.M. Christenson, T.J. Fahey, M.C. Fisk, C. Fuss, G.E. Likens, G. Lovett, L. Rustad, and P.H. Templer. 2018. Nitrogen oligotrophication in northern hardwood forests. Biogeochemistry 141:523-539. doi:10.1007/s10533-018-0445-y

Gundersen, P., B.A. Emmett, O.J. Kjønaas, C.J. Koopmans, and A. Tietema. 1998. Impact of nitrogen deposition on nitrogen cycling in forests: A synthesis of NITREX data. For. Ecol. Manage. 101:37–55. doi:10.1016/S0378-1127(97)00124-2

Hayhoe, K., C.P. Wake, T.G. Huntington, L. Luo, M.D. Schwartz, J. Sheffield, et al. 2007. Past and future changes in climate and hydrological indicators in the US Northeast. Clim. Dyn. 28:381–407. doi:10.1007/s00382-006-0187-8

Högberg, M.N., P. Högberg, and D.D. Myrold. 2007. Is microbial community composition in boreal forest soils determined by pH, C-to-N ratio, the trees, or all three? Oecologia 150:590–601. doi:10.1007/s00442-006-0562-5

Hooker, T.D., and J.E. Compton. 2003. Forest ecosystem carbon and nitrogen accumulation during the first century after agricultural abandonment. Ecol. Appl. 13:299–313. doi:10.1890/1051-0761(2003)013[0299:FECANA]2.0.CO;2

Houlton, B.Z., C.T. Driscoll, T.J. Fahey, G.E. Likens, P.M. Groffman, E.S. Bernhardt, and D.C. Buso. 2003. Nitrogen dynamics in ice storm-damaged forest ecosystems: Implications for nitrogen limitation theory. Ecosystems 6:431–443. doi:10.1007/s10021-002-0198-1

Houlton, B.Z., S.L. Morford, and R.A. Dahlgren. 2018. Convergent evidence for widespread rock nitrogen sources in Earth's surface environment. Science 360:58–62. doi:10.1126/science.aan4399

Hunt, J.F., T. Ohno, and I.J. Fernandez. 2008. Influence of foliar phosphorus and nitrogen contents on chemical properties of water extractable organic matter derived from fresh and decomposed sugar maple leaves. Soil Biol. Biochem. 40:1931–1939. doi:10.1016/j.soilbio.2008.04.004

Irland, L.C. 1998. Ice Storm 1998 and the forests of the Northeast: A preliminary assessment. J. For. 96:32–40. doi:10.1093/jof/96.9.32

Jefts, S., I.J. Fernandez, L.E. Rustad, and D.B. Dail. 2004. Decadal responses in soil N dynamics at the Bear Brook Watershed in Maine, USA. For. Ecol. Manage. 189:189–205. doi:10.1016/j.foreco.2003.08.011

Johnson, D.W., W. Cheng, and I.C. Burke. 2000. Biotic and abiotic nitrogen retention in a variety of forest soils. Soil Sci. Soc. Am. J. 64:1503. doi:10.2136/sssaj2000.6441503x

Kahl, J.S., S.A. Norton, I.J. Fernandez, L.E. Rustad, and M. Handley. 1999. Nitrogen and sulfur input-output budgets in the experimental and reference watersheds, Bear Brook Watershed in Maine (BBWM). Environ. Monit. Assess. 55:113–131. doi:10.1023/A:1006162927311

Kahl, J.S., J.L. Stoddard, R. Haeuber, S.G. Paulsen, R. Birnbaum, F.A. Deviney, et al. 2004. Have US surface waters responded to the 1990 Clean Air Act Amendments? Environ. Sci. Technol. 38:484A–490A. doi:10.1021/es040686l

Kelly, C., S. Schoenholtz, and M.B. Adams. 2011. Contrasts in carbon and nitrogen ecosystem budgets in adjacent Norway spruce and Appalachian hardwood watersheds in the Fernow Experimental Forest, West Virginia. Fourth Interagency Conference on Research in the Watersheds. p. 26–30.

Kjønaas, O.J., and A.O. Stuanes. 2008. Effects of experimentally altered N input on foliage, litter production and increment in a Norway spruce stand, Gårdsjön, Sweden over a 12-year period. Int. J. Environ. Stud. 65:431–464. doi:10.1080/00207230701862322

Kopáček, J., B.J. Cosby, C.D. Evans, J. Hruška, F. Moldan, F. Oulehle, et al. 2013. Nitrogen, organic carbon and sulphur cycling in terrestrial ecosystems: Linking nitrogen saturation to carbon limitation of soil microbial processes. Biogeochemistry 115:33–51. doi:10.1007/s10533-013-9892-7

Kosiba, A.M., P.G. Schaberg, S.A. Rayback, and G.J. Hawley. 2018. The surprising recovery of red spruce growth shows links to decreased acid deposition and elevated temperature. Sci. Total Environ. 637–638:1480–1491. doi:10.1016/j.scitotenv.2018.05.010

Lawrence, G.B., M.B. David, S.W. Bailey, and W.C. Shortle. 1997. Assessment of soil calcium status in red spruce forests in the northeastern United States. Biogeochemistry 38:19–39. doi:10.1023/A:1005790130253

Lovett, G.M., M.A. Arthur, K.C. Weathers, R.D. Fitzhugh, and P.H. Templer. 2013. Nitrogen addition increases carbon storage in soils, but not in trees, in an eastern US deciduous forest. Ecosystems 16:980–1001. doi:10.1007/s10021-013-9662-3

Lovett, G.M., and C.L. Goodale. 2011. A new conceptual model of nitrogen saturation based on experimental nitrogen addition to an oak forest. Ecosystems 14:615–631. doi:10.1007/s10021-011-9432-z

Lovett, G.M., and S.E. Lindberg. 1993. Atmospheric deposition and canopy interactions of nitrogen in forests. Can. J. For. Res. 23:1603–1616. doi:10.1139/x93-200

Lu, M., Y. Yang, Y. Luo, C. Fang, X. Zhou, J. Chen, et al. 2011. Responses of ecosystem nitrogen cycle to nitrogen addition: A meta-analysis. New Phytol. 189:1040–1050. doi:10.1111/j.1469-8137.2010.03563.x

Magill, A.H., J.D. Aber, G.M. Berntson, W.H. McDowell, K.J. Nadelhoffer, J.M. Melillo, and P. Steudler. 2000. Long-term nitrogen additions and nitrogen saturation in two temperate forests. Ecosystems 3:238–253. doi:10.1007/

s100210000023
Magill, A.H., J.D. Aber, W.S. Currie, K.J. Nadelhoffer, M.E. Martin, W.H. McDowell, et al. 2004. Ecosystem response to 15 years of chronic nitrogen additions at the Harvard Forest LTER, Massachusetts, USA. For. Ecol. Manage. 196:7–28. doi:10.1016/j.foreco.2004.03.033
McLauchlan, K.K., L.M. Gerhart, J.J. Battles, J.M. Craine, A.J. Elmore, P.E. Higuera, M.C. Mack, B.E. McNeil, D.M. Nelson, N. Pederson, and S.S. Perakis. 2017. Centennial-scale reductions in nitrogen availability in temperate forests of the United States. Sci. Rep. 7:7856. doi:10.1038/s41598-017-08170-z
McNulty, S.G., J.L. Boggs, J.D. Aber, and L.E. Rustad. 2017. Spruce-fir forest changes during a 30-year nitrogen saturation experiment. Sci. Total Environ. 605–606:376–390. doi:10.1016/j.scitotenv.2017.06.147
McNulty, S.G., E.C. Cohen, J.A. Moore Myers, T.J. Sullivan, and H. Li. 2007. Estimates of critical acid loads and exceedances for forest soils across the conterminous United States. Environ. Pollut. 149:281–292. doi:10.1016/j.envpol.2007.05.025
Mineau, M.M., F.R. Fatemi, I.J. Fernandez, and K.S. Simon. 2014. Microbial enzyme activity at the watershed scale: Response to chronic nitrogen deposition and acute phosphorus enrichment. Biogeochemistry 117:131–142. doi:10.1007/s10533-013-9869-6
Minocha, R., S.A. Turlapati, S. Long, W.H. McDowell, S.C. Minocha, and P. Millard. 2015. Long-term trends of changes in pine and oak foliar nitrogen metabolism in response to chronic nitrogen amendments at Harvard Forest, MA. Tree Physiol. 35:894–909. doi:10.1093/treephys/tpv044
Mitchell, M.J., C.T. Driscoll, J.S. Kahl, G.E. Likens, P.S. Murdoch, and L.H. Pardo. 1996. Climatic control of nitrate loss from forested watersheds in the northeast United States. Environ. Sci. Technol. 30:2609–2612. doi:10.1021/es9600237
Mitchell, M.J., N.W.W. Foster, J.P.P. Shepard, I.K. Morrison, and L.K. Morrison. 1992. Nutrient cycling in Huntington Forest and Turkey Lakes deciduous stands: Nitrogen and sulfur. Can. J. For. Res. 22:457–464. doi:10.1139/x92-060
Moldan, F., and R.F. Wright. 2011. Nitrogen leaching and acidification during 19 years of NH_4NO_3 additions to a coniferous-forested catchment at Gardsjön, Sweden (NITREX). Environ. Pollut. 159:431–440. doi:10.1016/j.envpol.2010.10.025
Morrison, E.W., A. Pringle, L.T.A. van Diepen, and S.D. Frey. 2018. Simulated nitrogen deposition favors stress-tolerant fungi with low potential for decomposition. Soil Biol. Biochem. 125:75–85. doi:10.1016/j.soilbio.2018.06.027
Morse, J.L., J. Durán, F. Beall, E.M. Enanga, I.F. Creed, I.J. Fernandez, and P.M. Groffman. 2015. Soil denitrification fluxes from three northeastern North American forests across a range of nitrogen deposition. Oecologia 177:17–27. doi:10.1007/s00442-014-3117-1
Nadelhoffer, K.J., M. Downs, B. Fry, A.H. Magill, and J.D. Aber. 1999. Controls on N retention and exports in a forested watershed. Environ. Monit. Assess. 55:187–210. doi:10.1023/A:1006190222768
Nave, L.E., E.D. Vance, C.W. Swanston, and P.S. Curtis. 2009. Impacts of elevated N inputs on north temperate forest soil C storage, C/N, and net N-mineralization. Geoderma 153:231–240. doi:10.1016/j.geoderma.2009.08.012
Niu, S., A.T. Classen, J.S. Dukes, P. Kardol, L. Liu, Y. Luo, et al. 2016. Global patterns and substrate-based mechanisms of the terrestrial nitrogen cycle. Ecol. Lett. 19:697–709. doi:10.1111/ele.12591
Norton, S.A., I.J. Fernandez, J.S. Kahl, L.E. Rustad, T. Navrátil, and H. Almquist. 2010. The evolution of the science of Bear Brook Watershed in Maine, USA. Environ. Monit. Assess. 171:3–21. doi:10.1007/s10661-010-1528-y
Norton, S.A., J.S. Kahl, I.J. Fernandez, T. Haines, L. Rustad, S. Nodvin, et al. 1999. The Bear Brook Watershed, Maine (BBWM), USA. Environ. Monit. Assess. 55:7–51. doi:10.1023/A:1006115011381
Ohno, T., I.J. Fernandez, S. Hiradate, and J.F. Sherman. 2007. Effects of soil acidification and forest type on water soluble soil organic matter properties. Geoderma 140:176–187. doi:10.1016/j.geoderma.2007.04.004
Oulehle, F., K. Tahovská, T. Chuman, C.D. Evans, J. Hruška, M. Růžek, and J. Bárta. 2018. Comparison of the impacts of acid and nitrogen additions on carbon fluxes in European conifer and broadleaf forests. Environ. Pollut. 238:884–893. doi:10.1016/j.envpol.2018.03.081
Pardo, L.H., M.J. Robin-Abbott, M.E. Fenn, C.L. Goodale, L.H. Geiser, C.T. Driscoll, et al. 2011. Effects and empirical critical loads of nitrogen for ecoregions of the United States. Ecol. Appl. 21:3049–3082. doi:10.1890/10-2341.1
Patel, K.F., and I.J. Fernandez. 2018. Nitrogen mineralization in O horizon soils during 27 years of nitrogen enrichment at the Bear Brook Watershed in Maine, USA. Environ. Monit. Assess. 190:563. doi:10.1007/s10661-018-6945-3
Patel, K.F., S.J. Nelson, C.J. Spencer, and I.J. Fernandez. 2018a. Fifteen–year record of soil temperature at the Bear Brook Watershed in Maine. Sci. Data 5:180153. doi:10.1038/sdata.2018.153
Patel, K.F., S.J. Nelson, C.J. Spencer, and I.J. Fernandez. 2018b. Soil temperature record for the Bear Brook Watershed in Maine. PANGAEA Data Publisher for Earth and Environmental Science. doi:10.1594/PANGAEA.885860
Perakis, S.S., J.E. Compton, and L.O. Hedin. 2005. Nitrogen retention across a gradient of ^{15}N additions to an unpolluted temperate forest soil in Chile. Ecology 86:96–105. doi:10.1890/04-0415
Piatek, K.B., and M.B. Adams. 2011. Are nitrate exports in stream water linked to nitrogen fluxes in decomposing foliar litter? Proceedings of the 17th Central Hardwood Forest Conference. p. 153–163.
Roskoski, J.P. 1980. Nitrogen fixation in hardwood forests of the northeastern United States. Plant Soil 54:33–44. doi:10.1007/BF02181997
Salinger, M.J. 2005. Climate variability and change: Past, present and future– An overview. Clim. Change 70:9–29. doi:10.1007/s10584-005-5936-x
SanClements, M.D., I.J. Fernandez, and S.A. Norton. 2010. Soil chemical and physical properties at the Bear Brook Watershed in Maine, USA. Environ. Monit. Assess. 171:111–128. doi:10.1007/s10661-010-1531-3
Schleppi, P., F. Curtaz, and K. Krause. 2017. Nitrate leaching from a sub-alpine coniferous forest subjected to experimentally increased N deposition for 20 years, and effects of tree girdling and felling. Biogeochemistry 134:319–335. doi:10.1007/s10533-017-0364-3
Simon, K.S., M.A. Chadwick, A.D. Huryn, and H.M. Valett. 2010. Stream ecosystem response to chronic deposition of N and acid at the Bear Brook Watershed, Maine. Environ. Monit. Assess. 171:83–92. doi:10.1007/s10661-010-1532-2
Smithwick, E.A.H., D.M. Eissenstat, G.M. Lovett, R.D. Bowden, L.E. Rustad, and C.T. Driscoll. 2013. Root stress and nitrogen deposition: Consequences and research priorities. New Phytol. 197:712–719. doi:10.1111/nph.12081
Stoddard, J.L. 1994. Long-term changes in watershed retention of nitrogen- Its causes and aquatic consequences. Adv. Chem. 237:223–284. doi:10.1021/ba-1994-0237.ch008
Stone, M.M., M.S. Weiss, C.L. Goodale, M.B. Adams, I.J. Fernandez, D.P. German, and S.D. Allison. 2012. Temperature sensitivity of soil enzyme kinetics under N-fertilization in two temperate forests. Glob. Change Biol. 18:1173–1184. doi:10.1111/j.1365-2486.2011.02545.x
Szillery, J.E., I.J. Fernandez, S.A. Norton, L.E. Rustad, and A.S. White. 2006. Using ion-exchange resins to study soil response to experimental watershed acidification. Environ. Monit. Assess. 116:383–398. doi:10.1007/s10661-006-7462-3
Tatariw, C., J.D. MacRae, I.J. Fernandez, M.-C. Gruselle, C.J. Salvino, and K.S. Simon. 2018. Chronic nitrogen enrichment at the watershed scale does not enhance microbial phosphorus limitation. Ecosystems 21:178–189. doi:10.1007/s10021-017-0140-1
Templer, P.H., G.M. Lovett, K.C. Weathers, S.E. Findlay, and T.E. Dawson. 2005. Influence of tree species on forest nitrogen retention in the Catskill Mountains, New York, USA. Ecosystems 8:1–16. doi:10.1007/s10021-004-0230-8
Templer, P.H., M.C. Mack, F.S. Chapin, L.M. Christenson, J.E. Compton, H.D. Crook, et al. 2012a. Sinks for nitrogen inputs in terrestrial ecosystems: A meta-analysis of ^{15}N tracer field studies. Ecology 93:1816–1829. doi:10.1890/11-1146.1
Templer, P.H., R.W. Pinder, and C.L. Goodale. 2012b. Effects of nitrogen deposition on greenhouse-gas fluxes for forests and grasslands of North America. Front. Ecol. Environ. 10:547–553. doi:10.1890/120055
Thomas, R.Q., C.D. Canham, K.C. Weathers, and C.L. Goodale. 2010. Increased tree carbon storage in response to nitrogen deposition in the US. Nat. Geosci. 3:13–17. doi:10.1038/ngeo721
Tietema, A., A.W. Boxman, M. Bredemeier, B.A. Emmett, F. Moldan, P. Gundersen, et al. 1998. Nitrogen saturation experiments (NITREX) in coniferous forest ecosystems in Europe: A summary of results. Environ. Pollut. 102:433–437. doi:10.1016/S0269-7491(98)80065-1
Treseder, K.K. 2008. Nitrogen additions and microbial biomass: A meta-analysis of ecosystem studies. Ecol. Lett. 11:1111–1120. doi:10.1111/j.1461-0248.2008.01230.x
Van Diepen, L.T.A., S.D. Frey, C.M. Sthultz, E.W. Morrison, R. Minocha, and A. Pringle. 2015. Changes in litter quality caused by simulated nitrogen deposition reinforce the N-induced suppression of litter decay. Ecosphere 6:1–16. doi:10.1890/ES15-00262.1
Venterea, R.T., P.M. Groffman, M.S. Castro, L.V. Verchot, I.J. Fernandez, and M.B. Adams. 2004. Soil emissions of nitric oxide in two forest watersheds subjected to elevated N inputs. For. Ecol. Manage. 196:335–349.

doi:10.1016/j.foreco.2004.03.028
Vitousek, P.M., and P.A. Matson. 1984. Mechanisms of nitrogen retention in forest ecosystems: A field experiment. Science 225:51–52. doi:10.1126/science.225.4657.51
Wallenstein, M.D., S.G. McNulty, I.J. Fernandez, J. Boggs, and W.H. Schlesinger. 2006. Nitrogen fertilization decreases forest soil fungal and bacterial biomass in three long-term experiments. For. Ecol. Manage. 222:459–468. doi:10.1016/j.foreco.2005.11.002
Wang, Z., and I.J. Fernandez. 1999. Soil type and forest vegetation influences on forest floor nitrogen dynamics at the Bear Brook Watershed in Maine (BBWM). Environ. Monit. Assess. 55:221–234. doi:10.1023/A:1006134120951
Watmough, S.A., J. Aherne, C. Alewell, P. Arp, S. Bailey, T. Clair, et al. 2005. Sulphate, nitrogen and base cation budgets at 21 forested catchments in Canada, the United States and Europe. Environ. Monit. Assess. 109. doi:10.1007/s10661-005-4336-z
Weathers, K.C., S.M.S. Simkin, G.M. Lovett, and S.S.E. Lindberg. 2006. Empirical modeling of atmospheric deposition in mountainous landscapes. Ecol. Appl. 16:1590–1607. doi:10.1890/1051-0761(2006)016[1590:EMOADI]2.0.CO;2
Wexler, S.K., C.L. Goodale, K.J. McGuire, S.W. Bailey, and P.M. Groffman. 2014. Isotopic signals of summer denitrification in a northern hardwood forested catchment. Proc. Natl. Acad. Sci. USA 111:16413–16418. doi:10.1073/pnas.1404321111
Wuebbles, D.J., D.R. Easterling, K. Hayhoe, T. Knutson, R.E. Kopp, et al. 2017. Our globally changing climate. In: D.J. Wuebbles, D.W. Fahey, K.A. Hibbard, D.J. Dokken, B.C. Stewart, and T.K. Maycock, editors, Fourth National Climate Assessment, Volume I. Global Change Research Program, Washington, DC. p. 35–72.
Yanai, R.D., M.A. Vadeboncoeur, S.P. Hamburg, M.A. Arthur, C.B. Fuss, P.M. Groffman, et al. 2013. From Missing source to missing sink: Long-term changes in the nitrogen budget of a northern hardwood forest. Environ. Sci. Technol. 47:11440–11448. doi:10.1021/es4025723
Young, H., J.H. Ribe, and K. Wainwright. 1980. Weight tables for trees and shrubs in Maine. Univ. of Maine, Orono, ME.