Spring 1999

The long-term effects of disturbance on nitrogen cycling and loss in the White Mountains, New Hampshire

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THE LONG-TERM EFFECTS OF DISTURBANCE ON NITROGEN CYCLING AND LOSS IN THE WHITE MOUNTAINS, NEW HAMPSHIRE

BY

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DISSERTATION

Submitted to the University of New Hampshire
in Partial Fulfillment of
the Requirements for the Degree of

Doctor of Philosophy

in

Natural Resources

May, 1999

UMI Number: 9926019

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Ann Arbor, MI 48103

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ACKNOWLEDGMENTS

With much gratitude, I thank John Aber for accepting me as a fledgling masters student six years ago, and for providing unqualified support, opportunity, and enthusiasm ever since. He and the rest of my committee, Andy Friedland, Jim Hornbeck, Bill McDowell, and Tom Lee have offered challenging discussions across community and ecosystem ecology and biogeochemistry. Particular thanks go to Bill McDowell and Jim Hornbeck for encouragement and analytical advice, and to Dan Zarin for thoughtful discussions on ecosystems, history, and how they fit together. I am extremely grateful to Peter Vitousek for encouraging the effort to revisit his work and for generously donating all of the original data.

Many individuals assisted in collecting the soil, foliage, and streamwater samples that made this thesis possible, including: Ruth Bristol, Deirdre Cunningham, Gary Filgate, Jennifer Jenkins, Maggie Lefer, Allison Magill, Erin Maiden, Jim Muckenhoupt, Steve Newman, Scott Ollinger, Jennifer Pett-Ridge, and Laura Stone. Particular thanks go to my father, Bruce Goodale, for time spent swatting black flies while learning to dbh and GPS. Further thanks go to Gloria Quigley, Matt Kizlinski, Jennifer Pontius, Maggie Lefer, Jane Hislop and Jeffrey Merriam for much-needed analytical assistance. Robert Down, Steve Newman, and Mary Martin patiently provided GIS tutorials, and helped bring century-old canvas maps into the digital age. Steve Fay and others at the White Mountain National Forest Headquarters provided access to historical records and encouragement for the research as a whole.

I have appreciated the unique experience of working on the MAP-BGC project with Scott Ollinger, Mary Martin, Marie-Louise Smith, and Rich Hallett. This thesis contributes a small piece to an effort designed to link AVIRIS remotely sensed imagery with land-use history and biogeochemical cycling.

I have greatly benefited from time spent sharing an office with Glenn Berntson, who opened the doors to statistics and other wonders, and with Jennifer Jenkins, whose organization I admire and whose company I enjoyed. Thanks too to Scott Ollinger for evolving discussions on the roles of species, physiology, nutrient cycling, and time.

A number of sources provided financial support for this work, particularly the NASA Global Change Fellowship Program. EPA grant #825865, NASA-TECO grant #NAG5-3527, and a Cooperative Agreement from the USDA Forest Service, Northeastern Research Station to W.H. McDowell funded additional portions of this research.

Finally, I thank both my parents, Bruce and Patricia Goodale, for their support, and I thank Mark Riddell for both field and laboratory assistance, and for patience, encouragement, and endurance during our respective four-year quests for knowledge.

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ABSTRACT

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NITROGEN CYCLING AND LOSS IN THE WHITE MOUNTAINS, NH

by

Christine L. Goodale

University of New Hampshire, May, 1999

Theories of nitrogen retention suggest that N cycling and loss should increase with ecosystem successional age and with chronic N deposition over time (N saturation). These factors both affect northeastern U.S. forests, most of which receive elevated rates of N deposition and have experienced past disturbances by wind, logging, fire, or agriculture. This work examined the long-term (80-110 year) effects of land-use history on nitrogen cycling and loss in the White Mountains, New Hampshire. Historical land-use maps were used to identify a network of watersheds and plots containing burned, logged, or old-growth forests. Nitrate-N fluxes from old-growth watersheds exceeded those from historically disturbed watersheds, yet losses from all forested watersheds were low (≤ 2.1 kg ha^{-1} yr^{-1}). Land-use history did not affect DON losses, which comprised 28-87% of total N fluxes, and increased with losses of dissolved organic carbon and with conifer forest cover. Old-growth stands had lower soil C:N ratios and twice the nitrification rates of historically disturbed stands. Nitrification increased as soil C:N ratio decreased, and stream nitrate concentrations increased with soil nitrification. Theories of N retention were further tested by remeasuring 28 streams sampled 23 years previously. Mean nitrate concentrations declined by 68%; calcium, by 28%; magnesium, by 26%; and

sulfate, by at least 22%. Nitrate concentrations declined in all streams, but old-growth watersheds had higher nitrate concentrations than successional watersheds in both years. Sulfate and base cation deposition have decreased since the 1970s but N deposition has not. Climate variability and its effects on biotic N retention may be responsible for the low nitrate losses observed in all streams, overriding expected increases due to chronic N deposition or forest aging. Century-old disturbances influenced spatial patterns of C:N ratio and nitrate production and loss, but climate may control temporal patterns of nitrate loss on the scale of months to decades. If the current, low losses of N are due to a high capacity to absorb N, forest ecosystems may continue to take up N for decades to centuries before reaching late-stage N saturation; if due to interannual climate variability, large losses of nitrate may occur much sooner.

CHAPTER 1

EFFECTS OF DISTURBANCE HISTORY ON ORGANIC AND INORGANIC NITROGEN LOSSES IN THE WHITE MOUNTAINS, NEW HAMPSHIRE

Abstract

Theories of nitrogen retention suggest that N losses should increase with increasing successional age, and with chronic N deposition over time; neither theory includes a specific time frame, nor addresses the role of losses of dissolved organic nitrogen (DON). This study examined the long-term (84-110 year) effects of land-use history in regulating the loss of both nitrate and DON from historically logged, burned, and old-growth forests in the northeastern United States. Monthly stream samples were collected from a network of watersheds with known land-use histories. Differences in stream nitrate concentrations drove seasonal differences in total N concentrations and spatial differences in modeled N flux. Mean (± SD) NO_3^--N loss from the old-growth watersheds (1.4 ± 0.6 kg ha^{-1} yr^{-1}) exceeded that from the historically logged and burned watersheds (0.3 ± 0.3 kg ha^{-1} yr^{-1}), demonstrating that the effects of disturbance can persist for 80-110 years. Successional status did not affect DON losses, which averaged 0.7 (± 0.2) kg ha^{-1} yr^{-1}, and comprised 28-87% of total N losses. DON losses increased with dissolved organic carbon losses and with cover by conifer or mixed hardwood and conifer forest. Despite decades of chronic, elevated N deposition, nitrate concentrations and fluxes were low. Total N retention appeared to decrease with time since disturbance, but even the old-growth stands retained at least 70% of estimated N inputs. Losses of

DON accounted for only 3-11% of N deposition, but could be more important controls on N accrual in other systems. Understanding the cause for the high rates of N retention observed in the present study is important: if due to a presently unsaturated capacity to absorb N, ecosystems may continue to take up N for decades to centuries before reaching late-stage N saturation; if due to interannual climate variability, large losses of nitrate may occur much sooner.

Introduction

Streams draining temperate forests demonstrate a wide variety of patterns and magnitudes of nitrogen loss. Two biogeochemical theories suggest that these patterns are governed by varying degrees of biotic control of the cycling and loss of N over time. The first theory focuses on the role of ecosystem successional status or time since the last major disturbance, while the second theory focuses on the role of human-induced alteration of N cycles and cumulative atmospheric N deposition (N saturation).

More than two decades ago, Vitousek and Reiners (1975) proposed that biotic control over the loss of limiting nutrients changes over the course of succession. After a pulse of nutrient loss following disturbance (in many, but not all systems; Vitousek et al. 1982), early and mid-successional systems were expected to retain virtually all limiting nutrients made available through deposition and internal nutrient cycling. Over successional time, net ecosystem production (Odum 1969) and net biotic demand for growth-limiting nutrients were expected to gradually decline to zero in old-growth systems (Vitousek and Reiners 1975, Gorham et al. 1979, Reiners 1981). With no net uptake, nutrient losses were expected to balance nutrient inputs. Streams draining old-growth forests have frequently (Vitousek and Reiners 1975, Leak and Martin 1975, Silsbee and Larson 1982, Flum and Nodvin 1995), but not always (Hedin et al. 1995) been shown to have high nitrate concentrations relative to those draining successional forests.

Over the past decade, there has been increasing recognition of the human impact on the global N cycle (Vitousek et al. 1997). Human activities have more than doubled the pre-industrial rate of production of biologically active N compounds (Galloway et al. 1995). The eastern United States currently receives 4-10 times the N deposition expected under unpolluted conditions (Galloway et al. 1984, 1995). Even higher rates of N deposition have been reported in western Europe (Dise and Wright 1995) and at high elevations in the eastern U.S. (Lovett and Kinsman 1990, Miller et al. 1993). Aber et al. (1989) hypothesized that N saturation occurs incrementally as chronic, elevated rates of atmospheric N inputs gradually satisfy biotic demands, leading to increased rates of nitrification and nitrate leaching losses (i.e., decreased N retention). Four stages of N saturation were proposed for terrestrial systems, and Stoddard (1994) described four complementary stages for surface waters, marked by increasing concentrations of nitrate, first during the dormant season, and later during the growing season. Forests in the late stages of N saturation are expected to have essentially no biotic N retention, with N losses approximately balancing N inputs.

Nitrogen saturated forests have been reported in both the eastern and western United States (reviewed in Fenn et al. 1998) and in western Europe (e.g., Dise and Wright 1995, Gundersen et al. 1998a). Yet, forests receiving similar rates of N deposition can have markedly different rates of nitrate loss (e.g., Hornbeck et al. 1997, Dise et al. 1998a, Lovett et al. in press), and there is growing recognition of the combined roles of ecosystem successional status and N deposition rates in regulating N retention and loss (van Miegroet et al. 1992, Stoddard 1994, Hedin et al. 1995, Aber and Driscoll 1997, Fenn et al. 1998). Neither of the two hypothesis specifically quantifies the length of time required to satisfy biotic demand under natural or anthropogenically enhanced N deposition, nor do they consider losses of N in dissolved organic forms (DON). Ecosystem loss of DON has often been overlooked, even though it can account for the majority of the hydrologic N losses from unpolluted, old-growth systems (Sollins et al. 1980, Hedin et al. 1995) and aggrading systems in regions of elevated N deposition (Lajtha et al. 1995, Currie et al. 1996).

This study specifically addressed the long-term (84-110 year) role of land-use history in regulating the loss of both nitrate and DON from historically disturbed and old-growth forests in the northeastern United States. I hypothesized that losses of total dissolved nitrogen (TDN) from old-growth forests would exceed losses from historically disturbed forests, and examined whether the mechanism of disturbance, logging or fire, impacted current N losses in streams. Both concentrations and estimated fluxes of nitrate and DON were compared across the different disturbance histories in order to assess: 1) whether losses of DON varied seasonally or spatially, and whether chronic losses of DON could account for significant cumulative losses of N from forests; 2) whether historically disturbed forests had higher N retention than old-growth forests, and whether N retention approached zero in the old-growth forests; and 3) whether forests exposed to chronic N deposition are showing signs of N saturation.

I sampled streams from a network of watersheds in the White Mountain National Forest, New Hampshire (WMNF), a region which has received elevated rates of N deposition for more than three decades (Likens and Bormann 1995). The region has a relatively well-defined disturbance history, comprised largely of century-old logging and fire, with limited forest harvest continuing through the present.

Methods

Site Description

The White Mountain National Forest (WMNF) consists of approximately 3000 km^2 of northern hardwood, spruce-fir, and subalpine forest in north-central New Hampshire (43.8 - 44.6 °N, 71.0 - 72.0 °W). Northern hardwood forests (Bormann and Likens 1979) cover most lower slopes, yielding to spruce-fir forest at approximately 750 m. Subalpine balsam fir *(Abies balsamea)* increases in importance above 1200 m until treeline gives way to alpine tundra at approximately 1400 m (Leak and Graber 1974, Reiners and Lang 1979). Over forty peaks exceed 1200 m, but nearly 90% of the land area falls below 750 m. Bedrock geology generally consists of highly-metamorphosed Devonian aluminum schists or Mesozoic granites (Hatch and Moench 1984). Throughout much of the northern hardwood zone, haplorthods developed on the stony tills deposited by glaciation 14,000 years ago. Precipitation is distributed evenly throughout the year (Federer et al. 1990), and increases with elevation (Dingman et al. 1981), from approximately 100 cm per year in the lowlands (300 m) to a long-term average of 230 cm at the summit of Mt. Washington (1917 m) (National Weather Service data). Snowpacks accumulate in winter, and streamflows at the Hubbard Brook Experimental Forest in the southwestern White Mountains usually peak with April snowmelt (Federer et al. 1990). Wet deposition of inorganic N (NH_4^+-N + NO_3^--N) currently averages 6.2 - 9.0 kg ha^{-1} yr^-

^{1} at Hubbard Brook and nearby Cone Pond, with DON contributing an additional 1.4 - 2.4 kg ha^{-1} yr^{-1} (Hornbeck et al. 1997, Campbell et al., submitted).

Forest History

Pollen records of post-glacial forest history indicate that northern hardwood communities were established in their present form by 7000 years before present, while red spruce *(Picea rubens)* did not join balsam fir in the present spruce-fir forests until 2000 years before present (Spear et al. 1994). Fires occurred rarely over the past 7000 years (Spear et al. 1994), with return intervals estimated at 750 to 2500 years (Lorimer 1977, Fahey and Reiners 1981). Wind is generally considered to be the primary mechanism of natural disturbance, with estimated return times of approximately 1150 years for a major wind disturbance (Lorimer 1977, Bormann and Likens 1979).

Most of the White Mountain region was held in public ownership until 1867, when the state of New Hampshire sold large tracts of forest to timber and paper companies. Intense clearcutting and slash fires followed over the next several decades, peaking shortly after the turn-of-the-century (Chittenden 1904). These relatively synchronous, large-scale, human-induced disturbances differed greatly from the natural disturbance regime, and are unique in the forest history of the region. Public criticism of forest practices brought about the passage of the Weeks Act in 1911, which funded federal purchase of forest land. Prior to purchase (generally 1911-1939), foresters surveyed and mapped each forest tract to evaluate its worth. Many of these maps and accompanying reports still exist (USDA Forest Service, Laconia, NH), providing a relatively complete record of the spatial extent of logged, burned, second growth, and old-growth forest early this century.

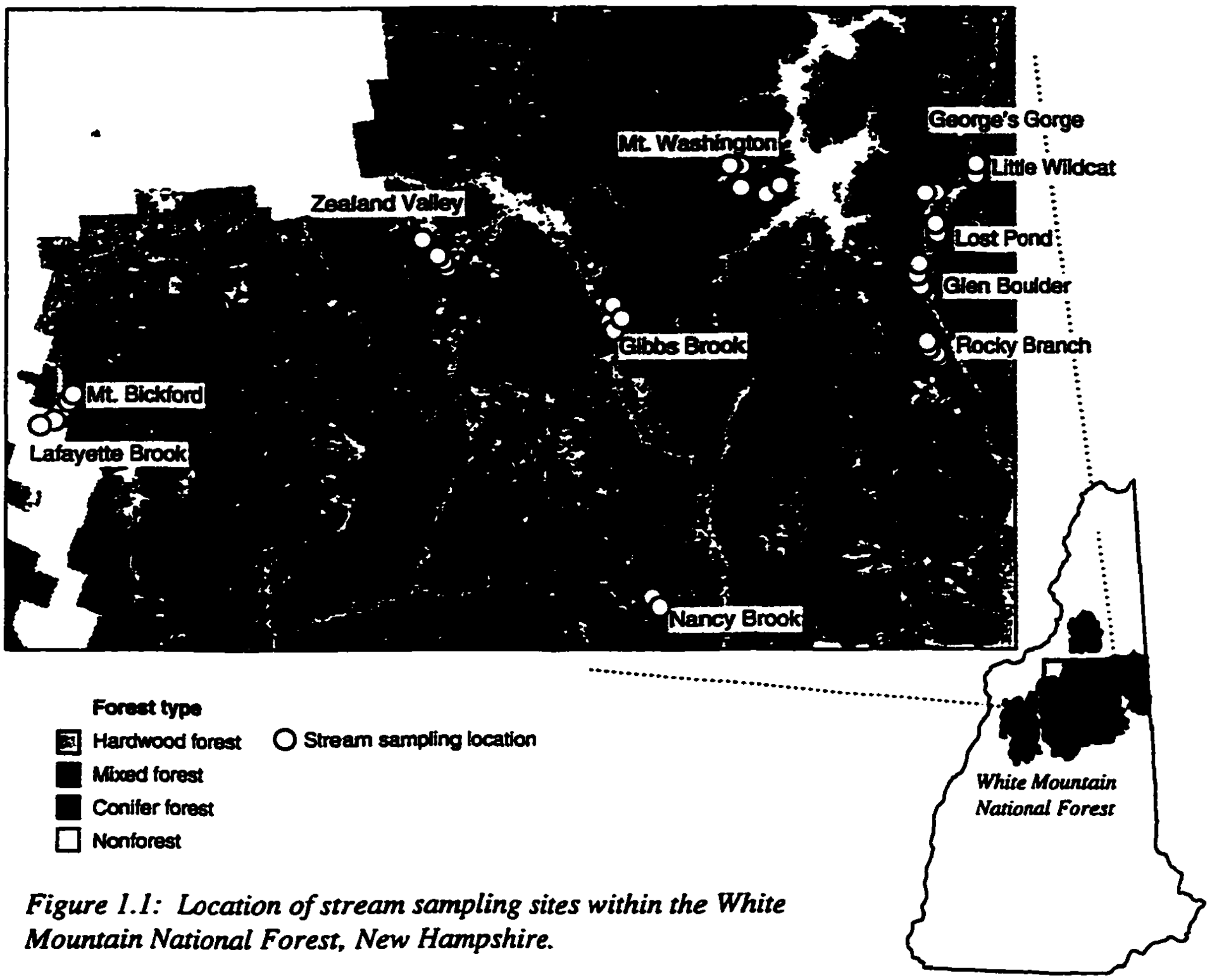

Figure 1.1: Location of stream sampling sites within the White Mountain National Forest, New Hampshire.

*Table 1.1: Watershed location, disturbance type and date, and number of streams sampled at each site. * streams sampled May - Sept. 1997; all others sampled Oct. 1996 - Sept. 1997.*

Disturbance	Year	Site	No. streams
Fire	1912-14	Rocky Branch Trail, Jackson	4
	c. 1903	George's Gorge, Pinkham Notch	2
	1903	*Mt. Bickford, Franconia	3
	1886	Zealand Valley	4
Logging	c. 1910	*Mt. Washington, Marshfield Station	2
	c. 1885	Little Wildcat	2
	c. 1885	Lost Pond Trail, Pinkham Notch	2
Old-growth		Glen Boulder, Pinkham Notch	3
		Gibbs Brook, Crawford Notch	4
		Lafayette Brook Scenic Area	2
		*Nancy Brook Research Natural Area	2
Partial landslide	1963	Slide Brook, Pinkham Notch	1
Subalpine/logged		Peabody Tributary, Pinkham Notch	1
Tundra/logged, tundra/subalpine	c. 1910	*Mt. Washington: Clay, Monroe, & Ammonoosuc Brooks	3

The historical survey maps were used to locate 30 streams in eleven areas each containing two to four small watersheds with relatively homogeneous land use histories (Figure 1.1). Forests on four sets of watersheds burned between 1886 and 1914; three sets of watersheds were logged heavily during this time period, and forests on four sets of watersheds have no record of logging or fire (Table 1.1). Five additional streams with watersheds containing non-forest (rock and tundra) or mixed land-use histories (partial landslide, mixed subalpine and logged, and mixed alpine and logged) were also sampled (Table 1.1). Historical records (Chittenden 1904, Belcher 1980, USDA Forest Service, Laconia, NH) and tree increment cores were used to estimate disturbance dates. Charcoal fragments were found in soils at all four of the historically burned sites. Of the old-growth sites, the Gibbs Brook (Foster and Reiners 1983) and Nancy Brook (Oosting and Billings 1951, Leak 1975) sites are previously documented old-growth stands, while the New Hampshire Natural Heritage Inventory identified old-growth northern hardwood stands in the Lafayette Brook Scenic Area (Sperduto and Engstrom 1993). The historical survey maps and stand compartment records (USDA Forest Service, Conway, NH) indicated that the Glen Boulder site contains old-aged mixed northern hardwoods with no record of major disturbance.

__Sample Collection and Analysis__

Streamwater samples were collected monthly from October 1996 to September 1997, except at the Nancy Brook, Mt. Washington, and Mt. Bickford sites, which were sampled from May to September 1997 (Table 1.1). The Nancy Brook and Mt. Washington sites were difficult to access in winter, and the Mt. Bickford site was added late in the study. Samples were collected over a two-three day period in the middle of each month. Samples were not collected in December, 1996; for annual flux calculations, December concentrations were estimated as the mean of November 1996 and January 1997 values. Similarly, missing chemistry for two streams added in January or February was approximated from other streams at the same site.

Samples were collected in 250 mL acid-washed (10% HCl) and well-rinsed high-density polyethylene (HDPE) bottles, which were rinsed with streamwater before collection. After collection, samples were refrigerated until processing the following day. An Orion® combination electrode was used to measure the pH of 30 mL aliquots at room temperature. The remaining sample was vacuum filtered through ashed (1 hour at 425 °C) Whatman GF/F glass fiber filters into acid-washed, rinsed sidearm flasks, and frozen until chemical analysis in polyethylene (NO_3^- and NH_4^+) or HDPE (DOC and TDN) 30 mL vials. After thawing to room temperature, NO_3^--N samples were filtered through 0.2 µm nominal pore size Acrodisks (Gelman Sciences) and analyzed with a Waters ion chromatograph and a Dionex AS4A column with micromembrane chemical suppression. Ammonium was measured with flow injection analysis using the automated phenolate method on a Lachet QuikChem® AE. Dissolved organic carbon (DOC) and TDN were estimated through separate analyses utilizing high-temperature (680 °C) catalytic (Pt) oxidation with a Shimadzu TOC 5000. DOC samples were acidified with 50 µL of 10% HCl and sparged prior to analysis. TDN was determined by oxidation and reaction with ozone to form NO_2, which was detected chemiluminescently with an Antek 720C N detector (Antek Instruments, Houston, TX) (Merriam et al. 1996). DON was calculated by difference as:

$DON = TDN - (NO_3^- \text{-N} + NH_4^+ \text{-N})$

The NO_3^- and DOC analyses were checked by repeated measures of independent commercial standards, which were always measured within 5% and 10% of their true values, respectively. TDN estimates were checked with NO_3^-, NH_4^+, glycine, and EDTA standards, which were usually measured within 10% of their true values. Duplicate measures of the same samples had coefficients of variation of less than 2% for NO_3^- and less than 8% for DOC, TDN, and NH_4^+.

Forest Cover Determination

Satellite-derived land cover information was obtained for the WMNF region from the New Hampshire Geographically Referenced Analysis and Information Transfer System (NH GRANIT, Durham, NH). NH GRANIT acquired Landsat TM data (pixel resolution of 30 m) for 1986-1990. Land cover was determined with a supervised classification of Landsat bands 3, 4, and 5, into ten cover types: wetlands, water, tundra, rock, agriculture, developed land, open/disturbed land, and hardwood, conifer, and mixed hardwood and conifer forest. Accuracy assessments indicated a state-wide average accuracy of 70% across all cover types (Rubin et al. 1993).

Watershed boundaries were delineated by hand on 7.5 minute (1:24,000) USGS topographic maps and digitized with Arc/Info version 7.1.1 (ESRI, Redlands, CA). Exact (SD < 5 m) sampling locations were identified with a Trimble Pro XR Global Positioning System (Sunnyvale, CA). These points were compared with digitized watershed boundaries to confirm that the correct starting positions had been identified on the USGS maps. The number of pixels of each cover type within each watershed were calculated by overlaying the digitized watershed boundaries and the land cover data with ERDAS

Imagine version 8.3 (Atlanta, GA). Although automated methods of watershed definition are available, I was unable to use these procedures because several watersheds were too small to delineate accurately with available digital elevation models (DEMs).

Hydrologic Fluxes

None of the sampled streams was gaged. 1996-7 streamflow was estimated at all of the sampled watersheds by using available climate data and a model of monthly carbon and water balances (PnET-II, Aber et al. 1995). The model utilizes monthly inputs of precipitation, temperature, and solar radiation to calculate evapotranspiration, snowpack development, and stream drainage, but does not consider flowpaths, groundwater, or soil hydrologic properties other than a specified water holding capacity and a fast or macropore flow rate. The model has captured temporal patterns in monthly streamflow at the Hubbard Brook Experimental Forest (Aber et al. 1995) and spatial patterns in mean annual runoff across New England (Ollinger et al. 1998).

*Table 1.2: Weather stations used to derive monthly precipitation estimates. Precip. is 1996-7 annual precipitation. * estimated; one or more month(s) missing data.*

Station	Start Date	Latitude	Longitude	Elev. (m)	Precip. (cm)
Benton	1940	44.03	-71.93	366	*99
Berlin	1931	44.48	-71.18	283	*111
Bethlehem	1931	44.28	-71.68	421	*94
Glencliff	1931	43.98	-71.90	329	*110
Jefferson	1940	44.37	-71.47	376	103
Lancaster	1931	44.48	-71.57	268	*101
Monroe	1931	44.32	-72.00	207	*100
Mt. Washington	1948	44.27	-71.30	1908	*315
North Conway	1974	44.05	-71.13	162	142
Pinkham Notch	1931	44.27	-71.25	613	*167
Plymouth	1931	43.78	-71.65	201	*121
Tamworth	1974	43.87	-71.30	241	154
York Pond	1931	44.50	-71.33	458	*126

Monthly precipitation data were obtained through 1997 from 13 U.S. cooperative or National Weather Service stations in the White Mountain region (Table 1.2). Monthly averaged maximum and minimum daily temperature data were obtained for the Bethlehem and Mt. Washington weather stations, and from four stations at the Hubbard Brook Experimental Forest (published through 1988 in Federer et al. 1990).

Monthly streamflow was estimated by running PnET-II for each 30 m pixel in each sampled watershed (a total of over 30,000 pixels). Pixel land cover and elevation were obtained from GIS overlays of the digitized watersheds and NH GRANIT land cover and DEM data layers. Precipitation and temperature were estimated at each pixel using its elevation and linear orographic factors derived for each monthly climate measurement from the available data. Precipitation generally increases and temperatures decrease with elevation in New England (Dingman 1981, Ollinger et al. 1995). In the present data set, elevation explained much of the spatial variation among monthly precipitation measurements (mean $R^2 = 0.67$), and nearly all of the spatial variation among monthly maximum and minimum temperature measurements (mean $R^2 = 0.98$ and 0.90, respectively). Pixels classified as conifer or hardwood were run with PnET-II parameter sets derived for spruce-fir or northern hardwood forests, respectively (Aber et al. 1995). Non-forest pixels (i.e., tundra, rock, or unvegetated open land) were assumed to have low precipitation interception (5%), low water holding capacity (2 cm), and no significant photosynthesis or transpirational demand. To verify PnET's estimates of streamflow, the model was run for several White Mountain streams with measured streamflow: the Ellis, Ammonoosuc, and East Branch of the Pemigewasset Rivers, all gaged by the USGS, and Watersheds 6 and 7 at the Hubbard Brook Experimental Forest.

The test watersheds were not digitized as described above; streamflow was simulated for each watershed's mean elevation and cover type.

Nitrogen Flux and Retention

Annual N fluxes (kg/ha) were estimated by multiplying measured stream N concentrations by estimated monthly streamflow, with monthly fluxes totaled for the water year. Net N retention was assessed across watersheds by comparing stream N flux (TDN_{out}) with simple estimates of N inputs (TDN_{in}). Net N retention was calculated as:

$$N\ Retention\ (\%) = (TDN_{in} - TDN_{out}) / TDN_{in} \times 100$$

Atmospheric deposition in mountainous regions can vary with elevation, slope, aspect, and canopy type, and can be extraordinarily difficult to estimate accurately (e.g., Lovett et al. 1997). Rough estimates of N inputs were calculated as the sum of a constant amount of dry deposition (1.8 kg/ha, derived from Ollinger et al. 1993), and wet deposition of constant concentration and varying precipitation amount (similar to Ollinger et al. 1993 and Miller et al. 1993). Precipitation amount was estimated with the elevation-based regressions, with an assumed N concentration of 0.60 mg/L. This value is the mean wet deposition TDN concentration measured at Hubbard Brook and Cone Pond during 1995-1997, consisting of 54% NO_3^--N, 28% NH_4^+-N, and 18% DON (Campbell et al., submitted). Solute concentrations in wet deposition might be expected to change with elevation, but a review by Lovett and Kinsman (1990) did not find any evidence to support consistent changes with elevation despite varying precipitation amount. Although dry deposition is likely to increase with elevation due to increased wind and other factors, it is difficult to consider these changes explicitly across multiple watersheds (Lovett and Kinsman 1990, Lovett et al. 1997). Mt. Washington and other mountain peaks receive substantial N inputs through cloud deposition (Lovett et al. 1982, Weathers et al. 1988, Lovett and Kinsman 1990, Miller et al. 1993), but this mechanism was not included because of limited data and the difficulty in assessing cloud exposure across the different watersheds.

Statistical Analyses

Differences in stream N concentration due to land-use history (historically burned, logged, or old-growth, Table 1.1) were determined by repeated measures analysis of variance, with a nested design of streams within site and sites within land-use. This approach considers the effects of land-use history at each measurement period after considering variability of streams within the same site, and the variability among sites with the same category of land-use history. Effects on stream N flux were assessed with similarly nested ANOVA. Scheffe's tests were used for all post hoc pairwise comparisons. Sites with only growing-season collections (Nancy Brook, Mt. Washington, and Mt. Bickford) were excluded from the ANOVAs, but were included in regression analyses of mean growing season chemistry and land cover. The three watersheds containing large areas of tundra and two watersheds with missing land cover data were not included in the forest cover analyses.

Results

Streamflow

The 1996-7 water year was wetter than normal: precipitation measured at the weather stations in Table 1.2 exceeded long-term (1971-1990) averages by 11% (range 6% - 20%). October and December, 1996 and July 1997 were each at least 25% wetter than average, while November 1996 and September 1997 were both drier than average.

Streamflow reflected the year's increased precipitation, with flow at Hubbard Brook and the USGS stations exceeding long-term averages by 10-20%.

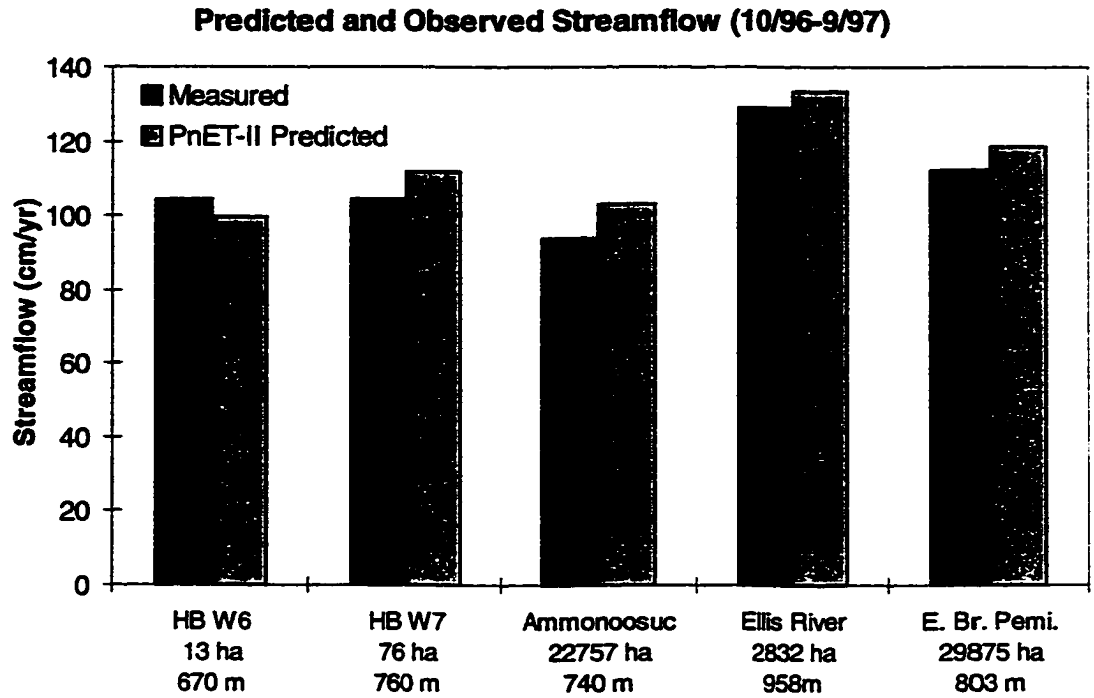

Figure 1.2: Predicted and observed annual (Oct. 1996 - Sept. 1997) streamflow for Watersheds 6 and 7, Hubbard Brook Experimental Forest, and three streams gaged by the USGS: the Ammonoosuc River at Bethlehem, the Ellis River above Jackson, and the East Branch of the Pemigewasset River at Lincoln. PnET-II was run for the mean watershed elevation (indicated).

The elevation-based regressions predicted annual precipitation within 2 and 21 mm of measured values at Hubbard Brook Watersheds 6 and 7, respectively. Modeled streamflow for the 1996-7 water year was within 4-10% of measured values at all five tested watersheds (Figure 1.2). The model generally reproduced monthly streamflow patterns at Hubbard Brook, although the elevation-based regressions underestimated December, 1996 precipitation at Hubbard Brook and PnET treated most of it as snow, leading to underestimates of December streamflow. The model captured the timing and magnitude of May snowmelt at the north-facing Watershed 7, but predicted a later snowmelt (April/May) than observed (April) at the south-facing Watershed 6 (Figure 1.3). The model does not consider the effects of aspect.

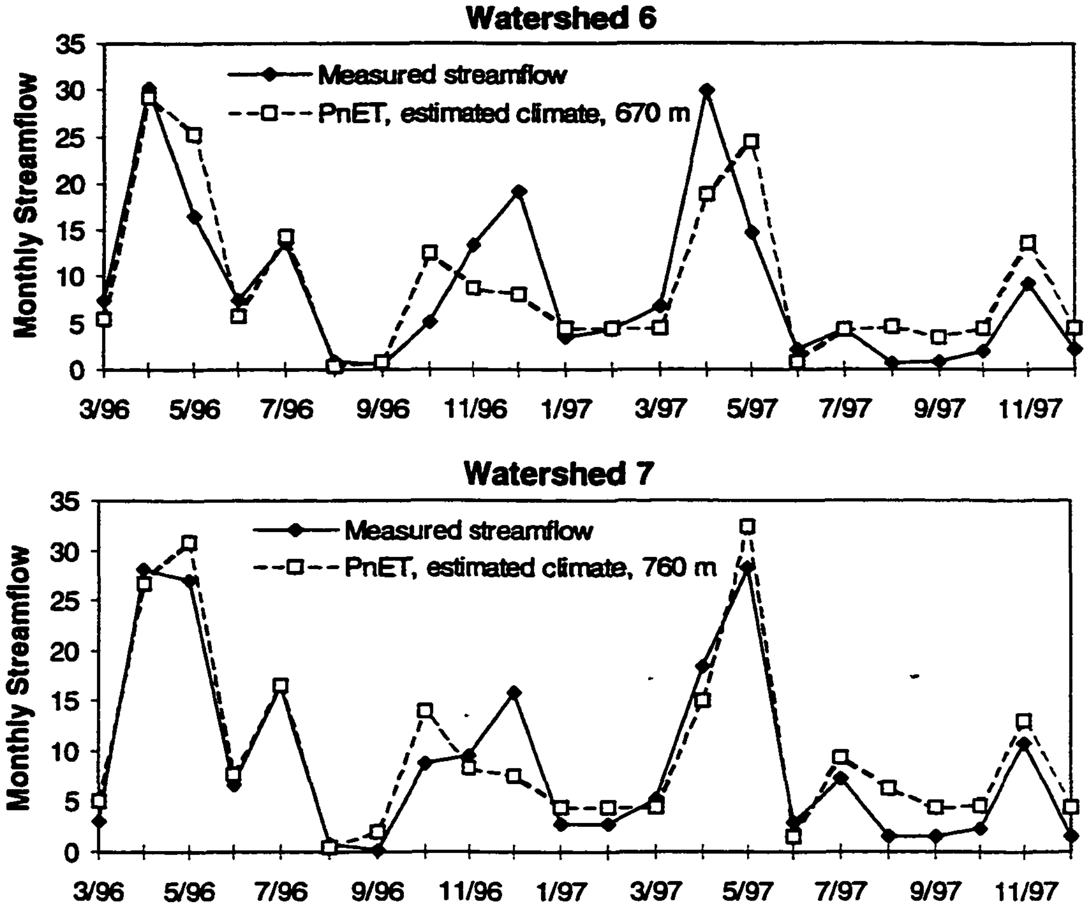

Figure 1.3: Predicted and observed monthly streamflow (cm) for Watersheds 6 and 7, Hubbard Brook Experimental Forest. PnET-II was run with mean watershed elevation, and the same elevation-driven climate inputs used for the other White Mountain watersheds.

The GIS analysis provided estimates of area, mean elevation, and forest cover for the sampled watersheds while the elevation-based regressions provided estimates of precipitation and hence wet N deposition by watershed (Table 1.3). Watersheds ranged in size from 4 to 232 ha, with a median size of 27 ha, and mean watershed elevation ranged from 559 to 1499 m with a median of 746 m. Differences among watersheds in estimated precipitation or N deposition were entirely due to differences in mean elevation.

Estimates of annual (wet + dry) N deposition ranged from 10.5 to 16.0 kg/ha, with a median value of 11.3 kg/ha. Differences among watersheds in modeled annual streamflow depended primarily on differences in estimated precipitation, although PnET-II did project slight variations in annual streamflow due to differences in land cover (northern hardwood, spruce-fir, or non-forest). The model predicted substantial differences among watersheds in monthly streamflow patterns due to variability in the timing and magnitude of snowmelt at different elevations: low-elevation watersheds were predicted to have melted out in April and May; higher watersheds in May, only; and the highest watersheds peaked in May but continued melting through June.

On average, the historically burned watersheds had less spruce-fir cover (9%) than the logged (27%) or old-growth (33%) watersheds. These differences in vegetation cover could be attributed to the differences in land-use history, or to slight differences in mean elevations. The historically burned watersheds were slightly lower in elevation (696 m) than the logged (778 m) or old-growth watersheds (804 m). All three land-use classes were represented by watersheds with a range of mean elevations (Table 1.3).

*Table 1.3: Watershed area, minimum and mean elevation, estimated mean precipitation, streamflow, and TDN deposition, and remotely sensed forest composition. Hard. = hardwood, Con. = conifer, Mix. = mixed hardwood/conifer, Non. = nonforest (rock, tundra, or wetland). Named streams are identified in parenthesis; unnamed streams, by letter. Precipitation estimates were derived from elevation-based regressions; streamflow, from PnET-II; and N deposition, as 0.60 mg/L TDN for wet deposition plus 1.8 kg/ha for dry. *indicates streams measured May - Sept., only.*

Site	Stream	Area (ha)	Elev (m) min.	Elev (m) mean	Precip. (cm)	Flow (cm)	N Dep. (kg/ha)	Vegetation Hard.	Vegetation Con.	Vegetation Mix.	Vegetation Non.
Historically burned											
Rocky Branch	a	16	509	637	106	95	10.8	100%			
1912-1914	b	44	519	694	156	103	11.2	100%			
	c	65	533	878	175	128	12.3	99%	< 1%	1%	
	d	17	540	708	158	105	11.2	100%			
George's Gorge	a (George's)	37	637	797	167	117	11.8	75%	15%	9%	1%
c. 1903	b	7	644	746	162	110	11.5	73%	20%	7%	
*Mt. Bickford	b	9	572	576	145	87	10.5	70%	17%	13%	
1903	c	27	570	618	149	93	10.7	72%	15%	13%	
	d	8	576	595	146	90	10.6	100%			
Zealand Valley	a	114	511	654	152	98	10.9	87%	9%	4%	1%
1886	b	4	522	574	144	88	10.4	76%	7%	17%	
	c	17	526	631	150	95	10.8	99%		1%	< 1%
	d (Hale)	225	531	878	176	127	12.3	54%	30%	15%	1%
Historically logged											
*Mt. Washington	b	22	888	949	182	136	12.7	13%	68%	10%	9%
c. 1910	c	4	873	890	176	127	12.3	5%	89%	6%	
Little Wildcat	a (L. Wild.)	175	533	872	174	125	12.2	38%	47%	14%	< 1%
	b	6	533	580	145	88	10.5	97%	2%	1%	
Lost Pond	a	12	615	659	153	99	10.9	63%	22%	15%	1%
	b	26	614	715	158	107	11.3	17%	62%	21%	1%
Old-growth											
Glen Boulder	b	36	543	905	177	132	12.4	75%	13%	6%	6%
	c	14	540	767	164	115	11.6	93%	1%	2%	6%
	d	42	534	929	180	133	12.6	61%	31%	6%	2%
Gibbs Brook	a (Gibbs)	218	640	966	183	135	12.8	7%	80%	11%	2%
	b	10	670	678	155	102	11.1	34%	52%	14%	
	c (Elephant)	123	583	930	180	132	12.6	9%	73%	10%	7%
	d	4	586	598	147	90	10.6	88%	2%	10%	
Lafayette Brook	a	9	587	604	147	92	10.6	n/a			
	b	7	588	611	148	93	10.7	n/a			
*Nancy Brook	a (Nancy)	215	733	995	186	140	13.0	9%	62%	23%	6%
	b	58	724	856	180	133	12.6	6%	45%	43%	5%
Alpine or mixed land-uses											
Slide Brook		186	558	1005	187	142	13.0	58%	26%	7%	9%
Peabody Trib.		68	632	963	183	135	12.7	32%	56%	9%	2%
*Mt. Washington	a (Clay)	232	876	1212	208	166	14.2	12%	44%	15%	30%
	d (Monroe)	105	983	1316	218	175	14.9	3%	63%	7%	28%
	e (Ammon.)	108	1049	1499	237	201	16.0	1%	25%	< 1%	74%

Nitrogen Concentrations

Differences in stream NO_3^--N concentrations drove seasonal differences in TDN concentrations, because NH_4^+-N concentrations were consistently low (6-12 µg/L, not shown) and DON concentrations showed no seasonal variation (Figure 1.4). Nitrate concentrations were strongly seasonal, peaking during the dormant season and dropping during the growing season. Streams draining old-growth stands had significantly higher NO_3^--N concentrations than those draining historically disturbed stands, particularly during the dormant season (Figure 1.5). Nitrate concentrations did not differ significantly between historically logged and burned sites in any month (Figure 1.5). DON and NH_4^+-N concentrations did not vary among the three land-use history classes in any month, so that the overall effects of land-use history on TDN concentration were controlled by differences in NO_3^--N concentration. Across all streams with annual chemistry (n = 25), DON contributed 57% (range 27-81%) of annual average TDN concentrations. DON contributed 57% and 69% of annual average TDN concentrations in the burned and logged stands, respectively, and 42% in the old-growth. Monthly DOC concentrations and DOC:DON ratios did not differ significantly by land-use history in any month, nor did they show any distinct seasonal patterns.

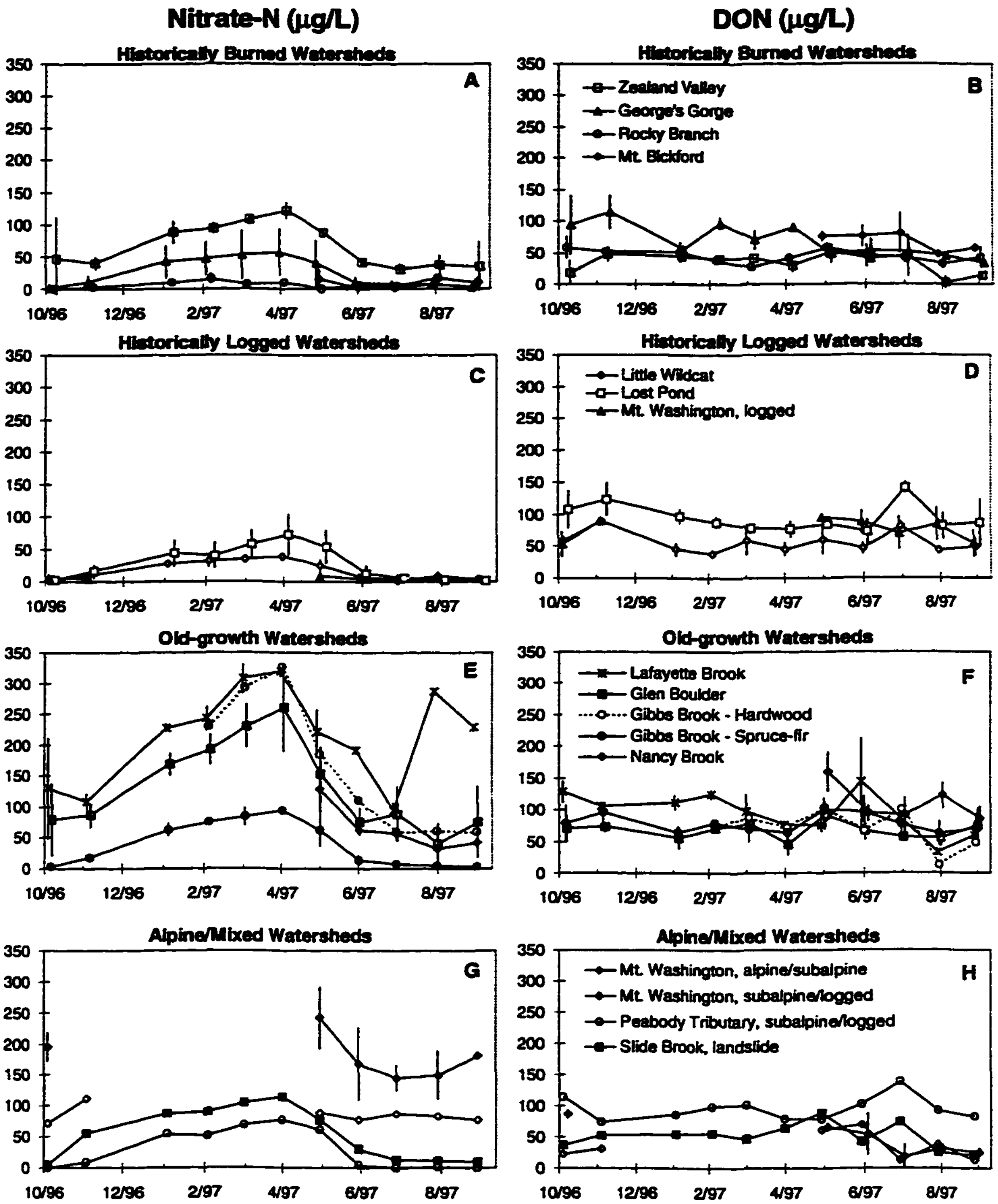

Figure 1.4: Monthly NO_3^--N and DON concentrations of historically burned (A, B), logged (C, D), old-growth (E, F), and alpine or mixed-history (G, H) sites. Plotted concentrations are the mean (SD) concentrations of sites identified in Tables 1.1 and 1.2.

Inconsistent with the overall trend of elevated NO_3^--N concentrations in streams draining old-growth forests, three of the four streams at the Gibbs Brook old-growth site had relatively low NO_3^--N concentrations (Figure 1.4e). These three watersheds all had more than 50% conifer cover, while the fourth stream, which drained a small, northern hardwood-dominated catchment, had NO_3^--N concentrations closely resembling the other old-growth sites (Figure 1.4e). Conifer-dominated old-growth catchments did not always have low NO_3^--N concentrations, however. Growing-season NO_3^--N concentrations from the Nancy Brook Research Natural Area, an old-growth red spruce *(Picea rubens)* site, exceeded those of the three conifer-dominated streams at Gibbs Brook, and approached the NO_3^--N concentrations of the other old-growth sites (Figure 1.4e).

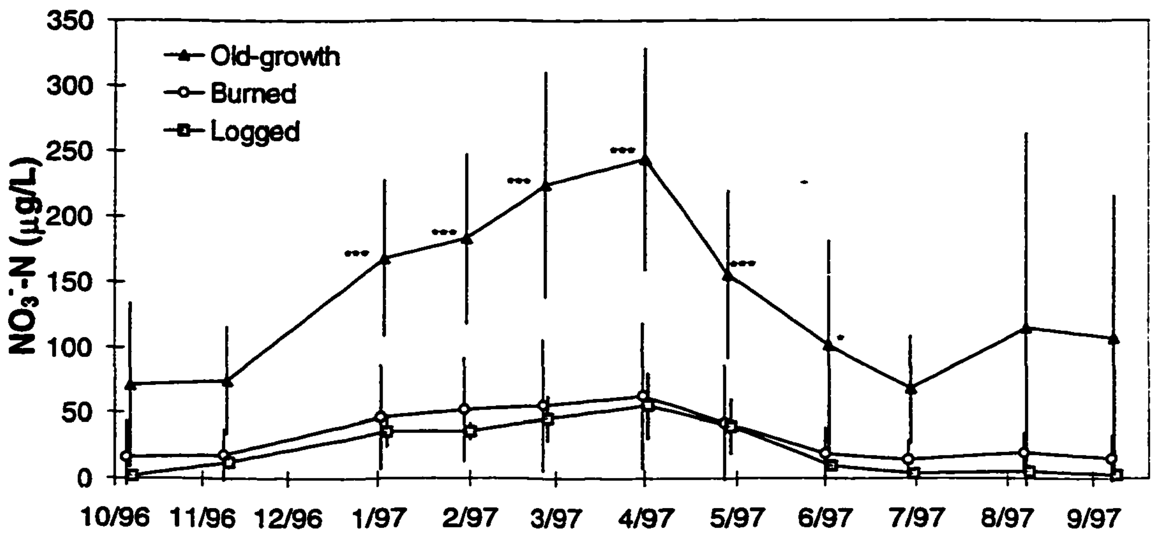

*Figure 1.5: Nitrate concentrations in streams draining old-growth, historically logged, or burned watersheds. Error bars are the standard deviation of NO_3^--N concentration among different sites. Analysis includes only streams with annual N measurements: burned, n = 3 sites (10 streams); logged, n = 2 sites (4 streams); old-growth, n = 3 sites (9 streams). * $P < 0.05$, *** $P < 0.0001$ for significantly different concentrations within a month.*

The highest NO_3^--N concentrations observed during the growing-season generally occurred in the high-elevation watersheds that contained large areas of alpine tundra: Monroe and Ammonoosuc Brooks on Mt. Washington (Figure 1.4g). The upper 30% of Clay Brook's watershed is above treeline, but much of the spruce-fir and mixed forest on the lower part of the watershed was logged in approximately 1910. Clay Brook's NO_3^--N concentrations fell intermediate to the relatively high concentrations of the high-alpine streams, and the near-zero values of nearby historically logged watersheds (Figure 1.4c).

Compared across all streams, mean growing-season DON and H^+ (estimated from pH) concentration increased with DOC concentration (Figure 1.6 a, b). Ammonoosuc Brook, a high-alpine stream on Mt. Washington, was an outlier in the relationship between DOC and H^+, with a much higher H^+ concentration than predicted by DOC concentration alone (circled, Figure 1.6b). This stream also had the highest mean growing season NO_3^--N concentration of all streams measured. With the exception of this stream, DOC concentrations explained 70% of the variability of mean H^+ concentrations. Both DOC and pH both corresponded with forest type: DOC concentrations increased and pHs decreased as the fraction of conifer or mixed hardwood/conifer forest increased (Figure 1.6 c, d).

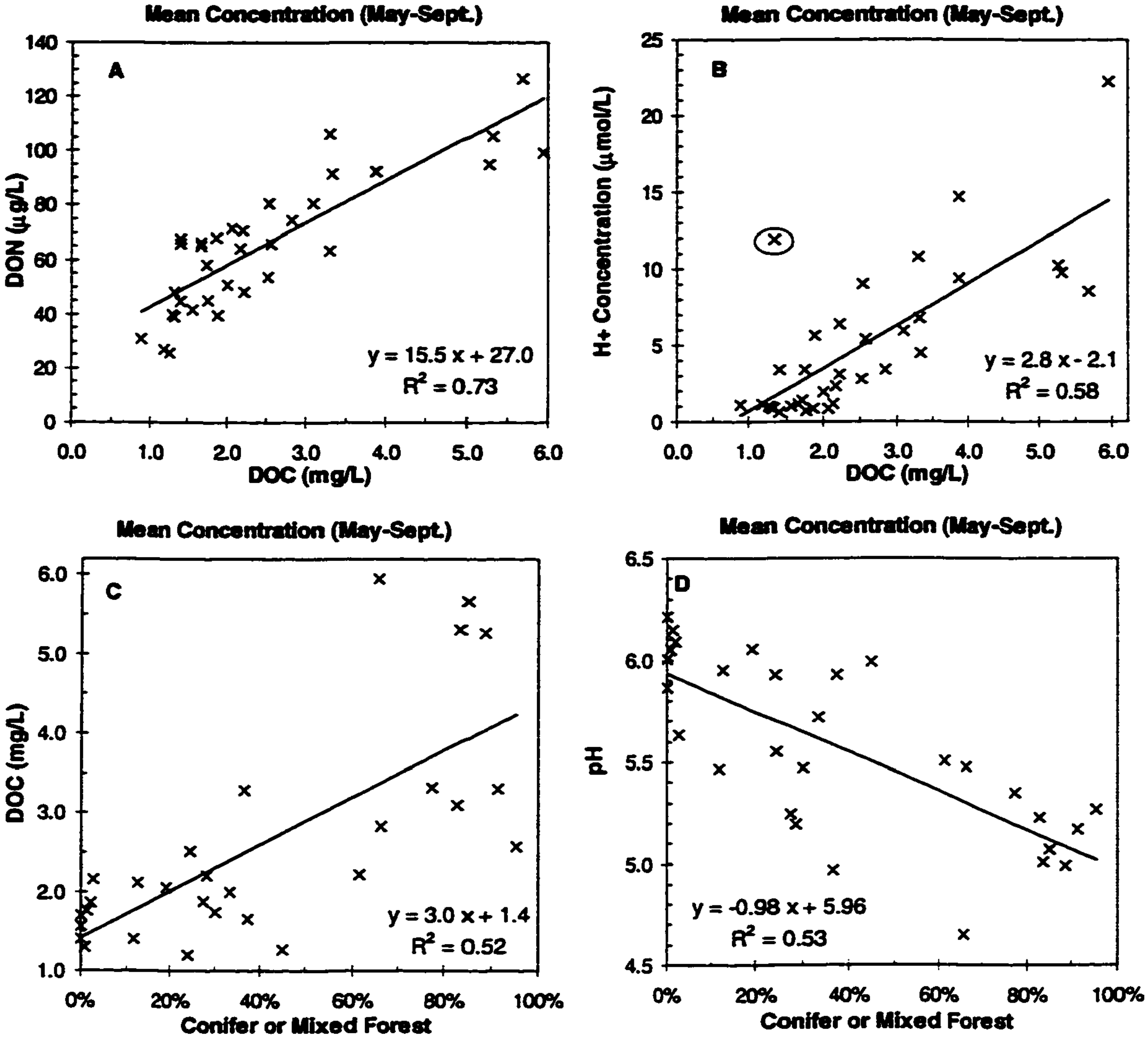

Figure 1.6: Mean (May - Sept.) DON (A) and H^+ (B) concentration as a function of DOC concentration, and mean DOC concentration (C) and stream pH (D) as a function of the fraction of watershed covered by conifer or mixed hardwood/conifer forest. The circled point in (B) is Ammonoosuc Brook, a high-alpine stream on Mt. Washington. $P < 0.001$ for all regressions; $n = 35$ (A, B) or 30 (C, D) streams.

Nitrogen Fluxes

Estimates of stream N flux were heavily influenced by the high-flow months of April and May (e.g., Figure 1.3), particularly when peak flows coincided with peak NO_3^--N concentrations in April (Figure 1.4). Differences among stream NO_3^--N concentrations overshadowed differences in the timing or magnitude of streamflow, such that differences among streams in predicted N flux resembled differences in mean N concentration. The median NO_3^--N loss from all of the forested catchments (n = 25) was 0.5 kg ha^{-1} yr^{-1}. Annual NO_3^--N losses ranged from < 0.1 kg ha^{-1} yr^{-1} in the Rocky Branch watersheds burned during 1912-1914, to nearly 3.0 kg/ha lost during May-Sept. alone from Ammonoosuc Brook, a stream draining Mt. Washington alpine tundra (Figure 1.7). If this stream had maintained its peak May, 1997 N concentrations all winter, annual NO_3^--N losses would have totaled approximately 4.0 kg/ha; annual TDN losses, 5.3 kg/ha. Across all forested catchments, the median DON loss was 0.7 kg ha^{-1} yr^{-1}, with a range from 0.4 kg ha^{-1} yr^{-1} in the Zealand Valley to 1.5 kg ha^{-1} yr^{-1} in Elephant Head Brook in the Gibbs Brook Research Natural Area (Figure 1.7). DON comprised 54% (range 28% to 87%) of annual TDN losses. DON losses were generally proportional to DOC losses, with a median DOC loss of 19.7 kg ha^{-1} yr^{-1}. DOC losses ranged from 9-13 kg ha^{-1} yr^{-1} in Zealand Valley streams to 62 kg ha^{-1} yr^{-1} in Elephant Head Brook and 67 kg ha^{-1} yr^{-1} in a tributary to the Peabody River.

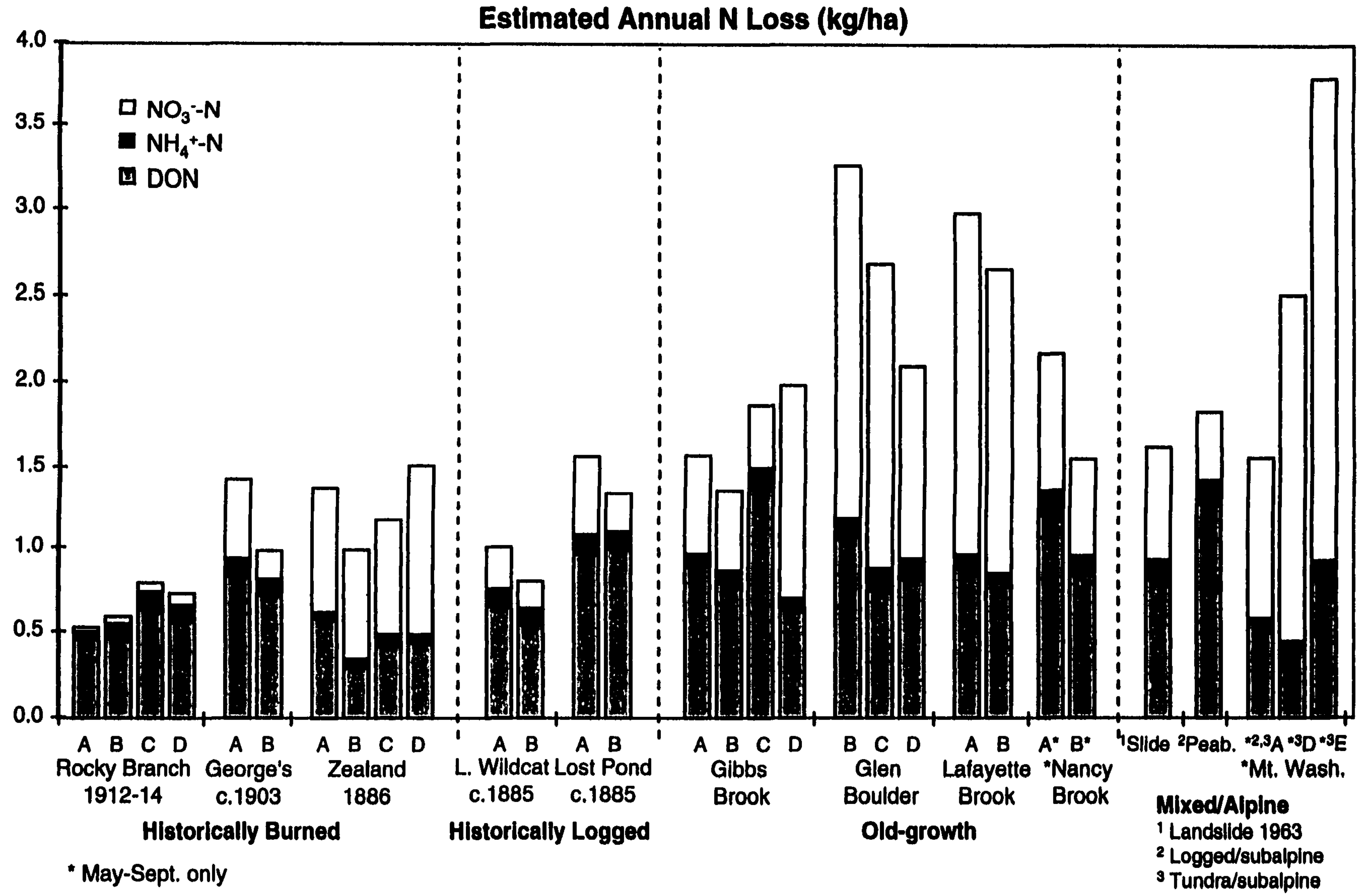

Figure 1.7: Estimated annual N flux (kg/ha) by stream. The Nancy Brook and Mount Washington sites (*) cover only May-Sept. 1997.

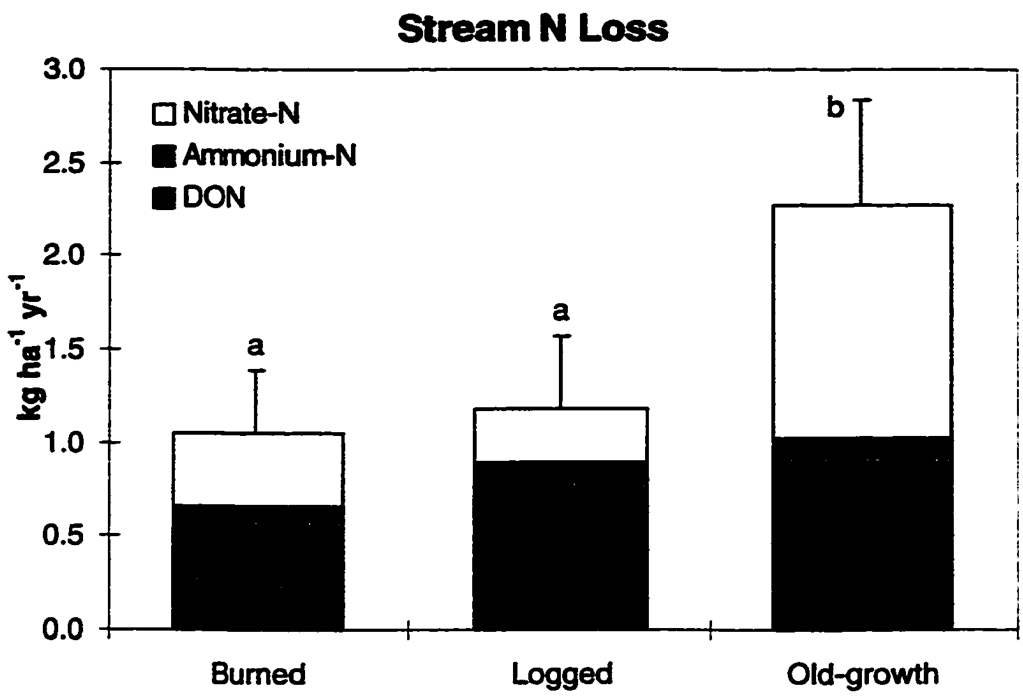

Figure 1.8: Mean stream NO_3^--N, NH_4^+-N, and DON flux (kg ha^{-1} yr^{-1}) by land-use history. Different letters indicate significantly different TDN fluxes. Error bars indicate the standard deviations among sites. Analyses include only streams with annual N measurements (n = 23).

Losses of TDN from old-growth watersheds were approximately double those from the historically logged or burned watersheds ($F = 6.46$, df = 2, 5, $P = 0.04$), primarily due to differences in nitrate loss (Figure 1.8). There were significant differences among sites within the same land use ($F = 7.4$, df = 5, 15, $P = 0.001$), in that Gibbs Brook had lower TDN losses than the two other old-growth sites. Annual NO_3^--N flux did not correlate with any watershed physical or chemical factors such as mean elevation (and streamflow), area, mean pH, or forest type, but increased marginally with decreasing DOC:DON ratio (Table 1.4). When only old-growth or mixed old-growth watersheds were considered, NO_3^--N losses increased more consistently with decreasing DOC:DON ratio (Figure 1.9). Several historically disturbed streams followed this pattern, but others had much lower nitrate fluxes than expected based on DOC:DON ratio alone.

Table 1.4: Correlation of annual nitrate and DON fluxes with stream chemistry and watershed features. Significant correlations are indicated in bold (Bonferroni-corrected alpha = 0.05 / 11 = 0.0045). Analyses include only streams with annual N measurements.

		Nitrate-N flux ($kg\ ha^{-1}\ yr^{-1}$)		DON flux ($kg\ ha^{-1}\ yr^{-1}$)	
	n	*R*	*P*	*R*	*P*
DON flux ($kg\ ha^{-1}\ yr^{-1}$)	25	0.19	0.36		
DOC:DON ratio	25	-0.48	0.01	0.43	0.03
Mean pH	25	0.03	0.88	-0.65	**0.0004**
Conifer cover (%)	23	-0.11	0.61	0.67	**0.0004**
Streamflow (cm/yr)	25	-0.02	0.91	0.51	0.009
Area (ha)	25	-0.11	0.60	0.10	0.64

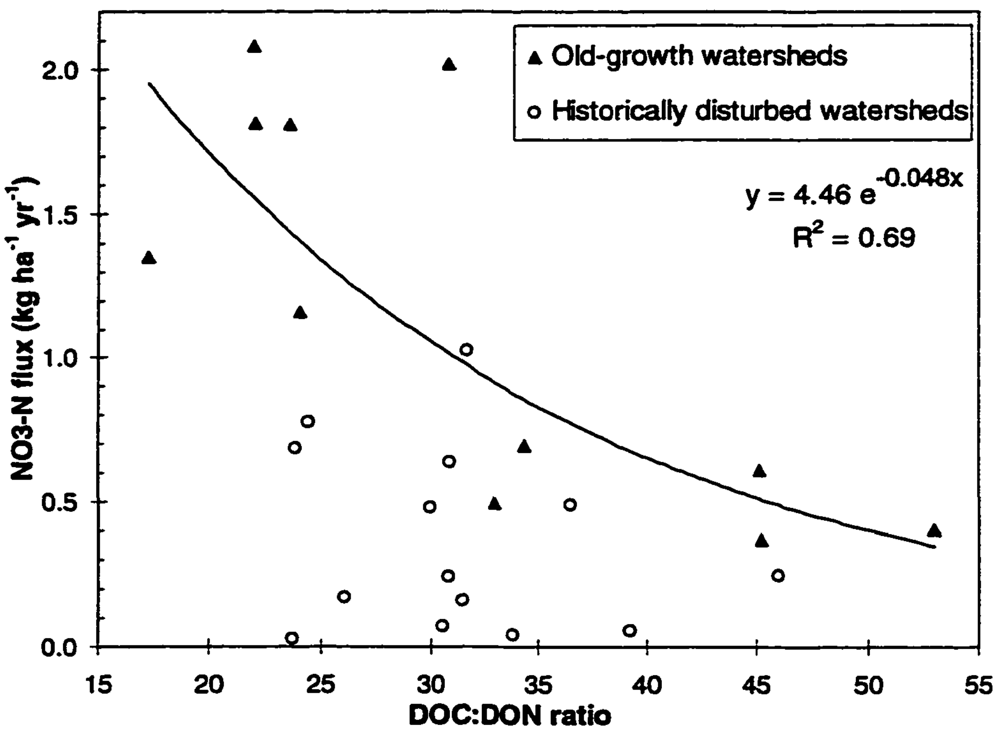

Figure 1.9: Annual NO_3^--N loss increased steeply with decreasing DOC:DON ratio when considered across streams draining old-growth forest (black triangles). Nitrate losses from historically disturbed stands (open circles) were often less than expected based on DOC:DON ratio alone. Curve and equation pertain to old-growth streams, only.

DON losses did not vary by land-use history ($F = 1.0$, df = 2, 5, $P = 0.43$), but instead followed patterns of DOC loss. Both DON and DOC fluxes increased with increasing cover by spruce-fir or mixed northern hardwood / spruce-fir forest (Table 1.4, Figure 1.6).

Nitrogen Retention

Net N retention appeared to decrease with increased time since disturbance (Figure 1.10), but the old-growth forests still retained at least 70% of estimated N inputs. Considering all forms of N loss, historically disturbed sites retained an average of 90% of added N while the old-growth sites retained 79% (Figure 1.10). Estimates of N retention would have been slightly higher (95% and 80% for disturbed and old-growth, respectively) if DON inputs and outputs had been ignored.

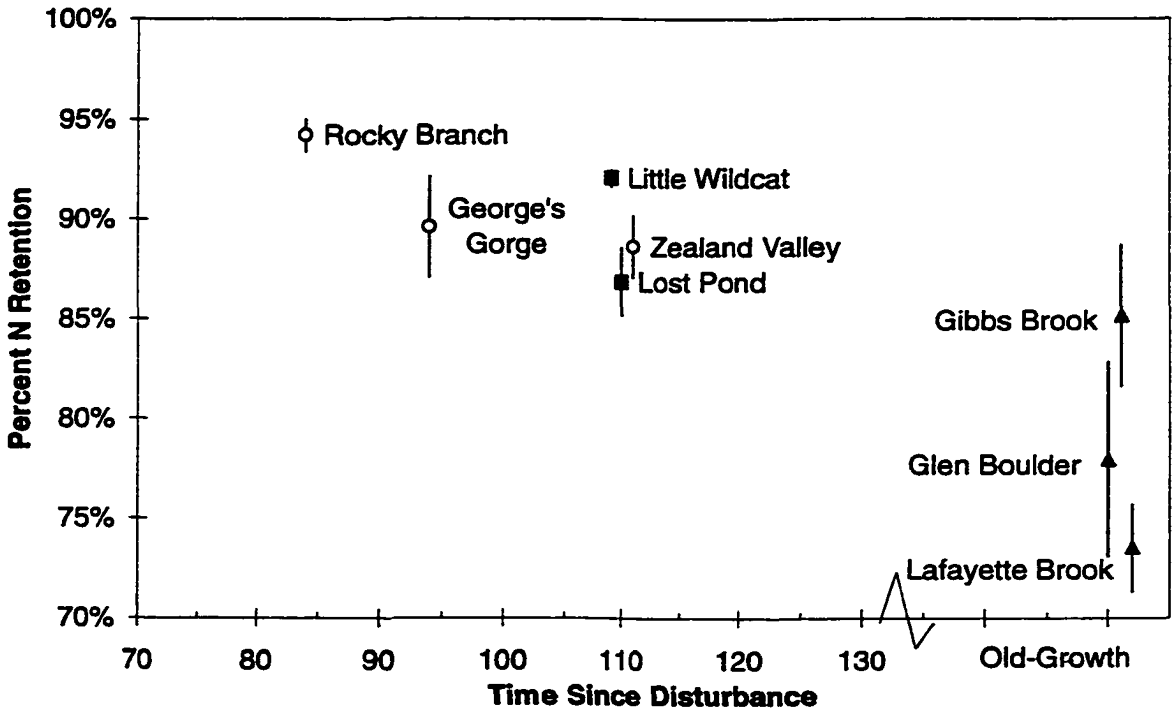

Figure 1.10: Net N retention (TDN_{in} - TDN_{out}) / TDN_{in} × 100 and time since disturbance. Error bars are the standard deviations among streams within sites.

Discussion

Constraints on N Flux Estimates

The reliability of my flux estimates depends on two main assumptions: first, that modeled estimates of streamflow were reasonable, and second, that stream chemistry on a single day in the middle of a month approximated the chemistry of the whole month. The modeled streamflow estimates depended on how accurately the watersheds were defined, how well the linear elevation factors represented spatial patterns of monthly precipitation, and how well the model predicted patterns of streamflow. Comparisons with Hubbard Brook suggested that the procedure performed well overall, but overestimated streamflow in October and late summer when NO_3^--N concentrations were low, while underestimating flow in December and, in some watersheds, in April when NO_3^--N concentrations were high (Figures 1.3, 1.4). Consistent errors in modeled streamflow of the magnitude observed at Watershed 6 (Figure 1.3a) would lead to underestimates of stream NO_3^--N flux by an average of 0.12 kg ha^{-1} yr^{-1} or 18% (range: < - 0.01 to -0.46 kg ha^{-1} yr^{-1}) for the streams in this study. The streamflow errors observed at Watershed 7 (Figure 1.3b) would underestimate N flux by an average of only 0.05 kg ha^{-1} yr^{-1} or 7% (range: + 0.05 to -0.22 kg ha^{-1} yr^{-1}). The larger potential errors occurred in streams with higher nitrate concentrations, while flux estimates changed little in streams with low nitrate concentrations.

The use of a single sample to represent a month's worth of stream chemistry can also introduce errors in estimated annual element flux. Concentrations of both nitrate (Murdoch and Stoddard 1992, Creed and Band 1998) and DOC (McDowell and Likens 1988, David et al. 1992) often increase at the start of high-flow events due to changes in hydrologic flowpaths and flushing of upper soil horizons; failure to sample high-flow periods may lead to underestimates of nitrate and DON flux. Most sampling dates in this study occurred during base flow conditions, although sampling dates in April and May coincided with high, but not peak snowmelt flows observed at Hubbard Brook Watershed 6. Sampling dates during the growing season coincided with both event (June and July) and base flow (August and September) conditions without discernible effects on NO_3^--N concentrations, although DOC and DON concentrations in some streams appeared to respond to the summer events (Figure 1.4). Swistock et al. (1997) report that monthly sampling of stream chemistry coupled with continuous streamflow measurements usually yielded estimates of annual nitrate flux within 10% (range -9.5% to + 41%) of estimates based on near-continuous chemical sampling, without any consistent bias. Eshleman et al. (1998) indicate that monthly chemistry and continuous streamflow measurements underestimate annual nitrate flux by an average of about 25% and 5%, respectively, for a highly responsive and less responsive stream in Virginia. Twenty-five percent underestimates of nitrate flux for the streams in this study amount to mean errors of 0.17 kg ha^{-1} yr^{-1} (range < -0.1 to -0.5 kg ha^{-1} yr^{-1}).

While there are uncertainties in the estimates of stream nitrate and DON flux (Figure 1.7), these estimates are similar to others reported for the northeastern U.S. Annual losses of DON from nine northern New England watersheds (including four at the Hubbard Brook Experimental Forest) during 1994-1997 ranged from 0.5 to 2.4 kg ha^{-1} yr^{-1}, while NO_3^--N losses ranged from < 0.1 kg ha^{-1} yr^{-1} to 2.1 kg ha^{-1} yr^{-1} (Campbell et al., submitted). Streams in the Catskill Mountains, New York, have higher mean NO_3^--N

(~3.5 kg ha^{-1} yr^{-1}) and similar mean DON fluxes (~0.7 kg ha^{-1} yr^{-1}) relative to those in this study (Lovett et al., in press).

Patterns of Dissolved Organic Nitrogen Losses

DON production in the forest floor has been shown to peak in late summer and early fall at the Harvard Forest, Massachusetts (Currie et al. 1996, McDowell et al. 1998). However, stream DON concentrations in this study showed little seasonal variation (Figure 4), a result consistent with other observations of stream chemistry in northeastern North America (Creed and Band 1998, Lovett et al. in press, Campbell et al. submitted). Retention of dissolved organic matter in the mineral soil (McDowell and Wood 1984) likely dampens any seasonal signal of DON production.

Successional status did not affect losses of DON (Figure 1.8), which instead was related more strongly to DOC losses (Figure 1.6a) and factors controlling DOC production. At the Harvard Forest, forest floors in conifer stands produced more DOC and DON, often with higher DOC:DON ratios than did forest floors in adjacent hardwood stands (Currie et al. 1996). Simlarly, Lawrence et al. (1986) found higher DOC concentrations in streams draining high-elevation spruce-fir forests than in those draining mid- and low-elevation northern hardwood forests. These results are consistent with the observation that DOC concentrations increased with percent cover by conifer or mixed forest (Figure 1.6c). DOC flux has been shown to increase with soil C:N ratio (Aitkenhead and McDowell, in press). The observed effects of forest type on DOC loss may have related to the higher C:N ratios in foliar and woody litter of conifer forests relative to hardwoods (Hobbie et al. 1992, Arthur et al. 1993).

Nitrate Loss and C:N Ratio

While DON fluxes appear to be generally controlled by factors governing DOC flux, nitrate fluxes were more closely controlled by forest successional status, although both forms of N loss may relate to soil C:N ratio. Nitrification and nitrate leaching losses generally increase as soil C:N decreases (Harmsen and Van Schreven 1955, van Miegroet et al. 1992, Gunderson et al. 1998b, Dise et al. 1998b, Chapter 2), and stream nitrate fluxes have been shown to correlate with decreasing C:N of dissolved organic matter (DOC:DON) in both Finland (Kortelainen et al. 1997) and New England (Campbell et al., submitted). While streams draining the old-growth stands in this study generally followed this relationship, several streams draining historically disturbed stands had much lower nitrate fluxes than predicted based on DOC:DON ratio alone (Figure 1.9). The ratio of DOC:DON may not always be analogous to soil C:N, in that DOC:DON ratios are frequently much higher than soil C:N ratios, likely reflecting a source other than simple solubilization from bulk humus (Currie et al. 1996). DOC:DON values observed in this study (mean = 32, range 17-53,), generally exceeded C:N ratios reported for soil organic matter at nearby sites (15-30; Huntington et al. 1988, McNulty et al. 1991, Chapter 2).

Nitrogen Losses and Forest History

The overall pattern of higher nitrate concentrations (Figure 1.5) and fluxes (Figure 1.8) in streams draining old-growth forests relative to successional forests is consistent with the hypothesis that old-aged systems should have lower net retention of N than aggrading successional systems (Vitousek and Reiners 1975). This study demonstrates that the effects of disturbance can persist for 80-110 years, despite chronic N deposition.

The two forms of disturbance, logging and fire, were expected to have had different effects on current N retention due to differences in disturbance intensity and N losses. Historical records did not provide sufficient detail to ascertain the relative intensity of logging or fire across sites, although the Zealand Valley fire received particular notoriety for its extent and severity (Chittenden 1904). Logged sites were presumed to have incurred N losses through bole removal and disturbance-induced nitrate leaching (e.g., Hornbeck and Kropelin 1982), while burned sites were presumed to have incurred these losses as well as losses of soil N through partial combustion of soil organic matter (e.g., Johnson et al. 1998). Most of the White Mountain fires 80-110 years ago were slash fires after logging (Chittenden 1904), and slash fires tend to burn hotter and longer than other surface fires, resulting in large losses of organic matter and N (Raison 1979). However, N retention did not appear to be affected by the form of historical disturbance, but by its date, with N losses increasing with time since disturbance (Figure 1.10). These results indicate either that the logging and fire both removed similar amounts of N, or that the differences in past N removal mattered less in controlling current N losses than current forest age.

Nitrogen losses from the conifer-dominated watersheds at Gibbs Brook, Crawford Notch, were lower than expected for an old-growth site (Figures 1.4e, 1.7). The Gibbs Brook watersheds have not been logged, but they may have experienced significant internal disturbance: several authors have noted extensive mortality of red spruce at the site in recent decades (Foster and Reiners 1983, Lawrence et al. 1997). Spruce mortality may have added large quantities of coarse woody debris to the forest floor, and placed much of the watersheds' living vegetation in an earlier successional state than presumed.

Nitrogen Retention, Saturation, and Succession

The strong seasonality of nitrate concentrations (Figure 1.4) suggested that substantial biotic uptake occurred in all of the forested watersheds during the growing season. Nitrate concentrations in the historically disturbed streams generally fit the model proposed by Stoddard (1994) for stage 0, the earliest stage of N saturation of surface waters. Stream NO_3^--N concentrations peaked during the dormant season at less than 150 μg/L, and generally fell to less than 10 μg/L during the growing season. The old-growth watersheds displayed the amplified seasonal cycle expected of stage 1, with peak steam NO_3^--N concentrations approaching those typical of wet deposition (~300 μg/L). The slightly elevated NO_3^--N concentrations evident during the growing season in the old-growth watersheds (~ 50 μg/L) could reflect reduced biotic uptake, leading toward the elevated nitrate concentrations in groundwater expected in stage 2. Similarly, streams draining the alpine tundra had elevated NO_3^--N losses during the growing season; yet concentrations remained less than 200 μg/L, suggesting a late stage 1 or very early stage 2. The old-growth watersheds did not demonstrate the relatively high, aseasonal nitrate concentrations that Vitousek and Reiners (1975) suggested were indicative of reduced biotic uptake in old age. Yet, recent measurements of nitrate in the same old-growth streams measured by Vitousek and Reiners (Chapter 3) suggest seasonal patterns similar to the old-growth streams in this study.

While TDN losses from the old-growth forests generally exceeded those from the historically disturbed forests, the old-growth forests still retained at least 70% of N inputs, far from the hypothetical net N retention of zero. Stream N fluxes could have been underestimated, but methodological errors do not explain differences of this magnitude. Failure to include cloud N deposition probably led to large underestimates of N inputs to some watersheds, implying that even higher rates of N retention likely occurred. Even the watershed dominated by alpine tundra retained at least 65% of N inputs, as calculated from liberal estimates of N loss (see results), and conservative estimates of N inputs (no cloud deposition). This estimate is consistent with Baron and Campbell (1997) and Kaste et al. (1997), who report 50-70% retention of N deposited on alpine tundra and rocky talus.

Why was N retention so high? Denitrification losses have been shown to be negligible in these and similar forests (Bowden and Bormann 1986, Bowden et al. 1991), so unmeasured gaseous N losses are not likely to explain the large difference between estimated N inputs and outputs. Imperfect knowledge of forest history may have missed episodes of past disturbance (e.g., the 1938 hurricane). Yet, cumulative leaching losses of nitrate following forest disturbance rarely exceed 60 kg N/ha (Martin et al. 1986), a loss that should have been replaced in less than a decade's worth of N deposition. Past disturbance could have placed the current old-growth forests into a slowly aggrading phase, rather than the transitional or degrading state expected to have no net N retention (cf. Bormann and Likens 1979, Peet 1992). Similarly, soils may still be accumulating N even if the vegetation is not, as exemplified by a 400-year-old Douglas-fir stand that accumulated N in soil organic matter and coarse woody debris even though it had stopped gaining biomass (Sollins et al. 1980).

Could chronic losses of DON preclude or substantially delay successional forests from accruing sufficient N in soils to lead to nitrification and nitrate loss? DON losses in the current study were quite low (mean = 0.7 kg ha^{-1} yr^{-1}), or approximately 3-11% of estimated TDN inputs. Similarly, approximately 12% of total N inputs to an old spruce-fir stand on Whiteface Mountain, New York, were lost below the rooting zone as DON (Friedland et al. 1991). Yet, DON losses can account for significant fractions of N inputs in some systems: DON losses in deep lysimeters (3-5 kg ha^{-1} yr^{-1}) accounted for approximately 30-60% of total N inputs to Harvard Forest and Cape Cod, Massachusetts forests (Lajtha et al. 1995, Currie et al. 1996). Half of the 2.0 kg ha^{-1} yr^{-1} of N deposited on an old Oregon forest was lost as DON in streamwater (Sollins et al. 1980), and stream losses of DON (2.3 - 4.6 kg ha^{-1} yr^{-1}) exceeded wet deposition of N to three Puerto Rican catchments (McDowell and Asbury 1994). Current losses of DON appear particularly large when compared with pre-industrial rates of N input. Chronic N deposition can enhance DON losses (Currie et al. 1996, McDowell et al. 1998), yet mean DON concentrations reported from pristine regions (133-217 μg/L; Hedin et al. 1995) slightly exceed those in this study. DON losses could perhaps have nearly balanced pre-industrial N inputs in some systems, substantially delaying N accumulation in soils to levels sufficient to induce nitrification and nitrate loss.

Interannual climate variability may also partially explain the current high N retention across all watersheds. At Hubbard Brook, retention of inorganic N within the same watershed varied erratically from less than 20% of wet N inputs in the early 1970's, to over 80% in the mid-1980s (Pardo et al. 1995), and 90% in the mid-1990s (Hornbeck et al. 1997, Campbell et al., submitted). Synchronous decreases in stream nitrate concentrations were observed across the White Mountain region during the 1990's (Driscoll et al. submitted, Martin et al., in prep., Chapter 3), yet no significant trends in N deposition have occurred over the past three decades (Likens and Bormann 1995, Driscoll et al., submitted). Using the PnET-CN model, Aber and Driscoll (1997) demonstrated that climate variability and its relative effects on N mineralization and plant uptake can explain much of the observed interannual variation in N retention at Hubbard Brook. The high N retention observed across all watersheds in this study may be due to particular climate factors in 1996-7 that favored plant and microbial uptake of N over N losses. Knowing the cause of the high rates of N retention is important: if due to a presently unsaturated capacity to absorb N, ecosystems may continue to take up N for decades to centuries before reaching late-stage N saturation; if due to climate variability, periodic episodes of large nitrate losses may occur much sooner.

CHAPTER II

LONG-TERM EFFECTS OF LOGGING AND FIRE ON NITROGEN CYCLING IN NORTHERN HARDWOOD FORESTS

Abstract

Most northeastern U.S. forests have been disturbed by wind, logging, fire, or agriculture over the past several centuries. These disturbances may have long-term impacts on forest carbon and nitrogen cycling, affecting forest vulnerability to N saturation and their future capacity to store C. I evaluated the long-term (80-110 year) effects of historical logging and fire on soil C and N pools, N turnover, and NO_3^- leaching in northern hardwood forests in the White Mountain National Forest, NH. Historical land-use maps were used to identify five areas each containing previously logged, burned, and relatively undisturbed (old-aged) forests. Forest floor masses in old-aged stands were smaller and soil C:N ratios were narrower than in historically burned or logged sites. The amount of N mineralized over 28 day laboratory incubations did not vary by land-use history, but mean (± SE) nitrification rates at old-aged sites (25 ± 2.9 kg N/ha) doubled those at burned (11 ± 2.9) and logged (13 ± 2.1) sites. Across all plots, nitrification increased as soil C:N ratio decreased, and NO_3^- concentrations in streamwater increased with soil nitrification. These results indicate that forest N cycling is affected by disturbances a century old. The increased nitrification at the old-growth northern

hardwood sites may have resulted from excess N accumulation relative to C accumulation in forest soils, due in part to chronic N deposition.

Introduction

Over the past several centuries, most northeastern U.S. forests have experienced human-induced disturbances such as forest harvest, fire, or agriculture (e.g., Chittenden 1904, Cronon 1983, Foster 1992, Foster et al. 1998). Successional forests in the eastern U.S. constitute a substantial sink in the global C budget (Birdsey et al. 1993, Dixon et al. 1994, Turner et al. 1995, Fan et al. 1998), a sink likely augmented by fertilization from atmospheric carbon dioxide or deposited nitrogen (Schimel 1995, Townsend et al. 1996, Houghton et al. 1998). These successional forests also provide a large sink for atmospheric N deposition, and different disturbance histories may partially explain the variety of forest responses observed to similar rates of N deposition (Aber and Driscoll 1997). Understanding the long-term impacts of historical disturbances on N cycling is important for predicting the potential amount of additional C and N that may be stored in northeastern forests.

Chronic N deposition may lead to N saturation, which is N availability in excess of plant and microbial demand, accompanied by elevated nitrification and NO_3^- leaching (Aber et al. 1989, Stoddard 1994). Field measurements have demonstrated that N losses do not necessarily increase directly with N inputs (van Miegroet et al. 1992, Magill et al. 1996, 1997, Gundersen et al. 1998a, Dise et al. 1998a), and watersheds with similar N inputs can have vastly different N outputs (Pardo et al. 1995, Hornbeck et al. 1997, Lovett et al., in press). These authors and others have inferred that past land-use history may partially regulate nitrate output. While the short-term impacts of disturbances on forest nutrient cycling have been well-studied (e.g., Likens et al. 1970, Raison 1979, Vitousek et al. 1979), long-term impacts are often overlooked. Modeling efforts suggest that disturbances may influence C and N cycling for hundreds of years (Aber and Driscoll 1997), but few studies have measured responses on this time scale, due in part to the difficulty of establishing land-use histories over such long periods. In this study, an unusually complete record of forest disturbance provided the opportunity to examine the effects of century-old fires and logging on current foliar chemistry, soil C and N pools, rates of N cycling, and nitrate losses in northern hardwood forests in the White Mountains, New Hampshire.

The White Mountain National Forest (WMNF) was established in 1911, largely in response to public outcries over widespread clearcutting and subsequent slash fires across the region during the preceding decades. These large-scale, relatively synchronous disturbances (Chittenden 1904) differed greatly from the natural disturbance regime, and are unique in the forest history of the region. Wind is the primary natural disturbance agent, and fires are rare in the northern hardwood zone (Lorimer 1977, Bormann and Likens 1979, Fahey and Reiners 1981, Spear et al. 1994). However, intense natural fires have occurred in limited areas in the past (e.g., Hornbeck et al. 1997). Forest clearance for cultivation or pasture was generally restricted to lowland valleys and floodplains, comprising a minor portion of the region's disturbance history.

The pulse of harvesting in the late 19th and early 20th centuries presumably removed large amounts of C and N from White Mountain forests. Current practices of forest harvest bring about large losses of N through both removal of biomass (Hornbeck and Kropelin 1982, Tritton et al. 1987) and induction of nitrate leaching (Vitousek et al.

1979, Martin and Pierce 1980, Hornbeck and Kropelin 1982). Historical forest practices in the White Mountains varied widely, from selective cutting of large-diameter spruce to intensive clearcutting on steep slopes. Slash was left on-site, and fires frequently struck cutover lands and spread to uncut forest. Period accounts often reported combustion of not only woody material, but soil organic matter (Chittenden 1904, and unpublished surveys, WMNF Headquarters, Laconia, NH). Nitrogen volatilization in fire corresponds directly with organic matter combustion (Raison 1979, Raison et al. 1985), but preferential combustion of fine litter rather than woody boles can lead to proportionately greater N than C losses (Johnson et al. 1998). System N losses from fire can be quite large, ranging from approximately 100 to 800 kg N/ha (Vose and Swank 1993, Johnson et al. 1998), and subsequent erosion can cause further losses of soil organic matter.

I hypothesized that past N removals through logging and fire created large N sinks relative to undisturbed forests. I expected that historically burned sites would be more severely impacted than logged sites, due to loss of soil N from the burned sites, and I predicted that N mineralization and nitrification, soil C and N pools, and nitrate leaching would increase from burned to logged to undisturbed sites.

Methods

Site Description

The WMNF covers 3000 km^2 in north-central New Hampshire (43.8 - 44.6 °N, 71.0 - 72.0 °W; Figure 2.1). Over forty peaks exceed 1200 m. These mountains largely consist of highly-metamorphosed Devonian aluminum schists or Mesozoic granites (Hatch and Moench 1984). Soils in the northern hardwood zone are primarily haplorthods, developed on stony glacial tills (Pilgrim and Peterson 1979). The Hubbard

Brook Experimental Forest, at 250 m elevation in the southwestern WMNF, receives an average of 1300 mm precipitation annually, and monthly mean temperatures range from -8.7 °C in January to 18.8 °C in July (Federer et al. 1990). Wet deposition of inorganic N (NH_4^+-N + NO_3^--N) averages 6.5 - 8.0 kg/ha at Hubbard Brook and nearby Cone Pond (Hornbeck et al. 1997). Temperatures decrease and precipitation and N deposition increase with elevation (Lovett and Kinsman 1990, Ollinger et al. 1993, 1995).

Some patches of old-aged northern hardwood forest escaped major damage from the logging, fires, and hurricanes of the past several centuries (e.g., Leak 1974, 1975, Foster and Reiners 1983) and forests have returned to nearly all of the disturbed areas. However, tree species' distributions and abundances have been substantially altered. Early and mid-successional species such as paper *(Betula papyifera)* and yellow birch *(B. alleghaniensis)*, aspen *(Populus spp.)*, and red maple *(Acer rubrum)* now dominate sites that previously may have supported late-successional American beech *(Fagus grandifolia)* and sugar maple *(A. saccharum)*, or on poorer sites, eastern hemlock *(Tsuga canadensis)* or red spruce *(Picea rubens)* (Leak 1991).

Site Histories

Old-aged, historically logged, and historically burned areas were located within each of five regions (Figure 2.1, Table 2.1). Historically burned and heavily cut stands were identified from published (Chittenden 1904, Belcher 1980) and unpublished records. At the time of purchase by the federal government – primarily 1911-1939 – the newly formed U.S. Forest Service surveyed and mapped each prospective forest parcel. These unpublished survey documents (WMNF Headquarters, Laconia, NH) identify both the type (northern hardwood, spruce-fir, or subalpine) and condition (burned, lightly cut, heavily cut, second growth, or virgin forest) of the forests at the time of purchase. Site disturbance histories were confirmed with inspection of stand structure, composition, and soils. All historically burned sites contained soil charcoal fragments, usually located at the contact between the forest floor and the mineral soil. Charcoal was not found at any of the other sites. Chittenden (1904), Belcher (1980), or survey reports confirmed the burn dates for all fires but that at the George's Gorge, Pinkham Notch site, which was approximated from tree increment cores. Information from the purchase surveys was used to constrain the dates of cutting in the logged stands.

Additional sources were used to identify the old-growth stands. As official or candidate Research Natural Areas, the Bowl (Leak 1974, Martin 1977, 1979) and Gibbs Brook (Foster and Reiners 1983) are well-known old-growth sites with documented forest histories. The New Hampshire Natural Heritage Inventory identified potential old-growth northern hardwood stands in the Lafayette Brook Scenic Area (Sperduto and Engstrom 1993) and in the Spruce Brook watershed of the Wild River valley (Engstrom and Sperduto 1994). The Glen Boulder site is a previously undocumented old-aged stand that was described as "virgin hardwoods and spruce" in its purchase survey in 1912, and compartment records indicate that no harvesting has occurred since then (USDA Forest Service, Conway, NH).

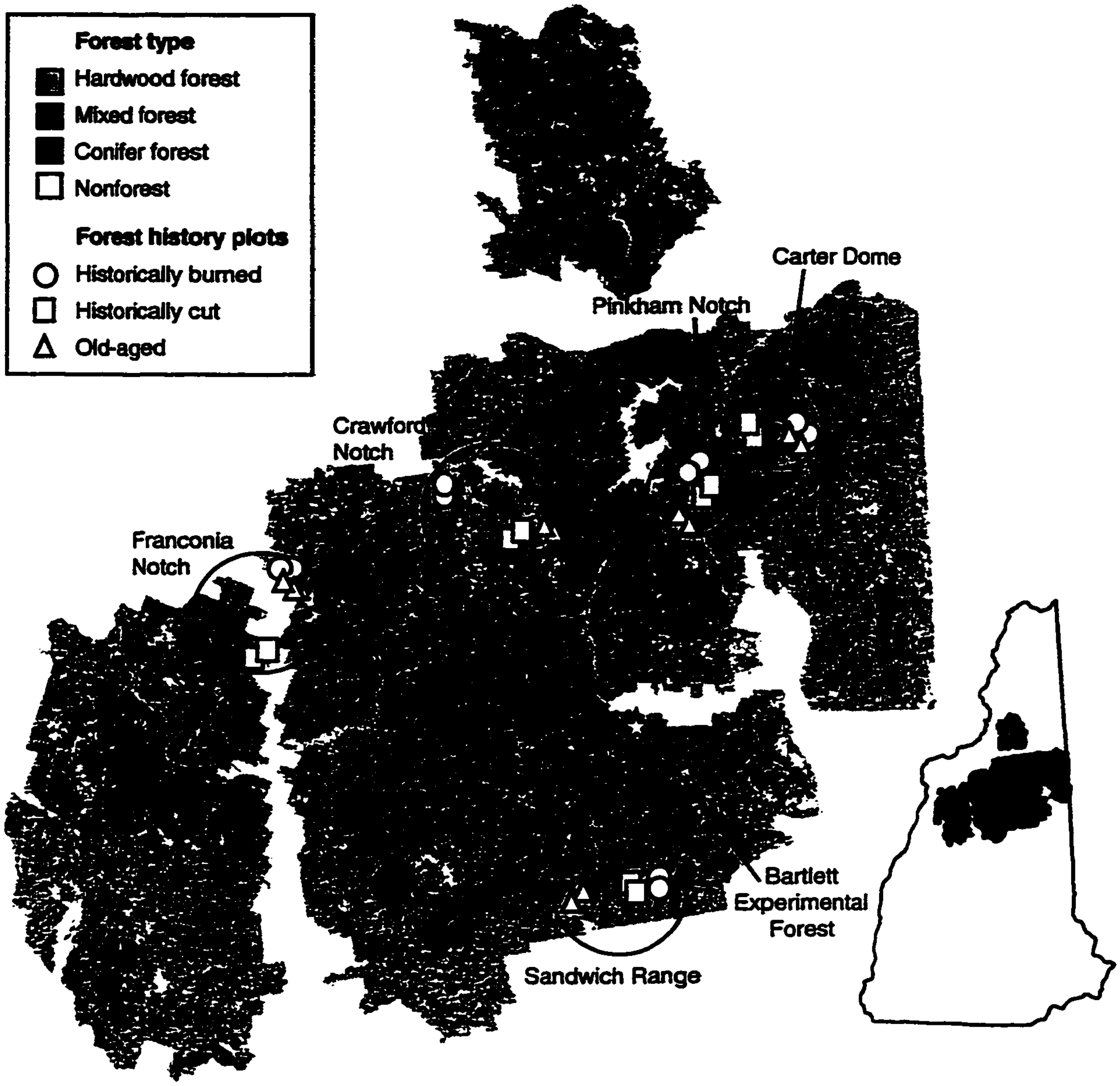

Figure 2.1: Soil sampling locations by land-use history (symbols) and region (circles) in the White Mountain National Forest, New Hampshire.

Table 2.1: Location and sampling date (1996) of soil collection sites. Dates of fires and possible dates of logging are indicated in parentheses. Asterisks indicate areas with streamwater collections.

Region	Date	Old-aged	Burned (date)	Logged (possible dates)
Sandwich Range	June 10-11	The Bowl	Mt. Chocorua (1915)	Mt. Paugus (c. 1915)
Pinkham Notch	June 17-18	*Glen Boulder	*George's Gorge (c. 1903)	*Lost Pond (1896-1915)
Crawford Notch	June 24-25	*Gibbs Brook	*Zealand Valley (1886)	Mt. Tom (1880-1915)
Franconia Notch	July 1-2	*Lafayette Brook	*Mt. Bickford (1903)	Cascade Brook (c. 1895)
Carter Dome	July 8-9	Spruce Brook	Wild River (1903)	Carter Dome Tr. (1896-1915)

Patterns of forest history could possibly have caused systematic biases in site selection. Red spruce was the dominant timber species at the last turn-of-the-century, and so historically logged sites currently occupied by successional hardwoods may have once supported spruce. Likewise, old-aged stands may have avoided cutting due to their lack of merchantable spruce at the time. Inherent site qualities such as slope, aspect, and soil mineralogy, texture, drainage, and thickness which govern red spruce distribution (Leak 1991) may also influence today's C and N pools and cycling rates. I tried to avoid confusing site history with other site factors by choosing logged sites that were marked as "cutover hardwood" on survey maps and using old-aged stands that were left uncut for reasons other than just species composition, such as watershed protection or inconvenient location within ownership boundaries. However, few old-growth stands exist, and it is difficult to be certain about the exact species composition of stands cut a century ago.

At each site, two 20 × 20 m plots were established on drained, mid-slope positions dominated by northern hardwoods. Recent canopy gaps were avoided. An altimeter was used to minimize elevation differences among plots within each region, and exact plot locations were later determined with a global positioning system. Plots ranged from 525 to 850 m elevation, with a mean of 650 m.

Soil Measurements

Net N mineralization and nitrification, soil C, N, and organic matter content, and soil pH were measured at all 30 plots (5 regions × 3 land-use histories × 2 plots/site). Net N mineralization and nitrification were estimated with 28 day laboratory incubations intended to mimic the buried bag method (Eno 1960, Nadelhoffer et al. 1983). Cores were incubated at a constant room temperature (mean = 21 °C, range = 19-23 °C) rather than fluctuating field temperatures. This laboratory method differed from laboratory potential mineralizations (e.g., Stanford and Smith 1972, Groot and Houba 1995, Knoepp and Swank 1995) in that soils were not sieved nor were soil moisture levels adjusted prior to incubation.

Soils were collected in early summer (Table 2.1) from three randomly chosen 5 × 5 m subplots within each 20 × 20 m plot. After removing recent litter (Oi), three pairs of 5.5 cm diameter soil cores were collected at each subplot. Each core was divided into forest floor (Oe + Oa) and mineral soil (0 to 10 cm) horizons, which were placed into separate polyethylene bags. Cores were refrigerated until return to the laboratory. One core from each pair was extracted within 24 hours of return, while the other was incubated in the dark for 28 days. Horizons were composited by subplot for all analyses by sieving through a 5.6 mm mesh sieve. Soils were weighed after sieving, and these weights were used to convert N turnover to units of area (kg/ha). This procedure assumes that soil components > 5.6 mm do not contribute significantly to extractable N concentrations. Soil moisture was determined as the weight loss of the composited soils dried at 105 °C for 48 hours. Ten grams of sieved, field-moist soil were extracted in 100 mL of 1 *M* KCl over 48 hours. Extracts were filtered through glass fiber filters (Gelman Sciences A/E) and frozen until later analysis. Extract NO_3^--N and NH_4^+-N concentration were analyzed colorimetrically with a Technicon TRAACS 800 Autoanalyzer using Technicon methods 782-86T (hydrazine reduction) and 780-86T (indophenol blue), respectively. N mineralization was calculated as: $([NO_3^- \text{-N}] + [NH_4^+ \text{-N}])_{incubated} - ([NO_3^- \text{-N}] + [NH_4^+ \text{-N}])_{initial}$, and nitrification was calculated as $[NO_3^- \text{-N}]_{incubated} - [NO_3^- \text{-N}]_{initial}$.

The laboratory incubation method was compared with annual measurements of N mineralization and nitrification at 14 plots at the Bartlett Experimental Forest in the WMNF. Nitrogen turnover was measured from June 1996 to May 1997 using the traditional buried bag technique (Ollinger et al., submitted). Additional soil cores were collected from these plots on July 19 and 23, 1996 for 28 day laboratory incubations and analyses as described above.

Composited soils from the initial cores were air dried and later analyzed for pH and carbon and nitrogen fractions. Soil pH was measured with an Orion glass pH electrode in a 1:10 g/g (forest floor) or 1:4 g/g (mineral soil) 0.01 *M* $CaCl_2$ slurry. Soil organic matter content was determined by loss-on-ignition at 500 °C for 5 hours. Carbon and nitrogen fractions were determined on dried, ground (Brinkman mechanized ceramic mortar and pestle) samples by combustion and gas chromatographic analysis using a Fisons NA 1500 Series 2 CHN analyzer.

Vegetation Measurements

On all 30 plots, diameter at breast height and species were recorded for all trees 9.5 cm dbh or greater. Foliage was collected from the mid- to upper canopy of up to three individuals of each canopy species during August, 1997, using 12-gauge shotguns and No. 4 steel shot. Foliage samples were air dried and ground with a Wiley mill to pass through a 1 mm mesh sieve. Nitrogen and lignin concentrations were determined with near-infrared reflectance spectroscopy (Wessman et al. 1988, McLellan et al. 1991, Bolster et al. 1996). On each plot, foliar chemistry was averaged by tree species, and then weighted by species' proportional basal areas to obtain plot-averaged foliar chemistry.

Stream Nitrate

Stream NO_3^--N concentrations were obtained for streams draining seven of the fifteen soil-collection sites (Table 2.1) as part of a larger survey of stream chemistry (Chapter 1). Two to four small streams were sampled at each site, and each stream's watershed fell wholly within the identified land-use history. Monthly streamwater grab samples were collected from Oct. 1996 until Sept. 1997 for all but the Mt. Bickford site, which was sampled from May - Sept. 1997. Its mean annual concentration was estimated from its growing season mean and the ratio of growing season : annual mean nitrate concentration in the other historically burned streams. Chapter 1 describes the sampling and analytical methods in detail.

Statistical Analyses

Laboratory estimates of N mineralization and nitrification at the Bartlett Experimental Forest were compared with annual *in situ* measurements using ordinary least-squares regression. The nitrification data were positively skewed and some values were slightly negative, so one kg/ha was added to all nitrification rates to allow log-transformation.

Nitrogen turnover, soil properties, and foliar chemistry were compared across the three land-use history categories (burned, logged, and old-aged) with analysis of variance. The ANOVAs were blocked by region (five levels, Table 2.1) as a fixed factor, and a land-use history × region interaction term was included. Scheffe's tests were used for post hoc comparisons among land-use histories. Relationships between plot-averaged soil properties were examined with correlation analyses, and stepwise multiple linear regressions were used to determine which among these properties best explained variability in net N mineralization and nitrification rates. Analysis of covariance was used to evaluate the effects of land-use history on nitrification rates while controlling for variation in sugar maple abundance.

Results

Methods Comparison

Estimates of net N mineralization (Figure 2.2a) and log-transformed nitrification (Figure 2.2b) from the 28 day laboratory incubations correlated strongly with annual field measurements at the Bartlett Experimental Forest. These results indicated that the laboratory incubations reliably captured field trends in N mineralization and nitrification. However, laboratory estimates of N turnover at the land-use history plots often exceeded values observed at any of the Bartlett plots, and so I cannot be certain that the strong correlations between field and laboratory trends persisted at the higher N turnover rates.

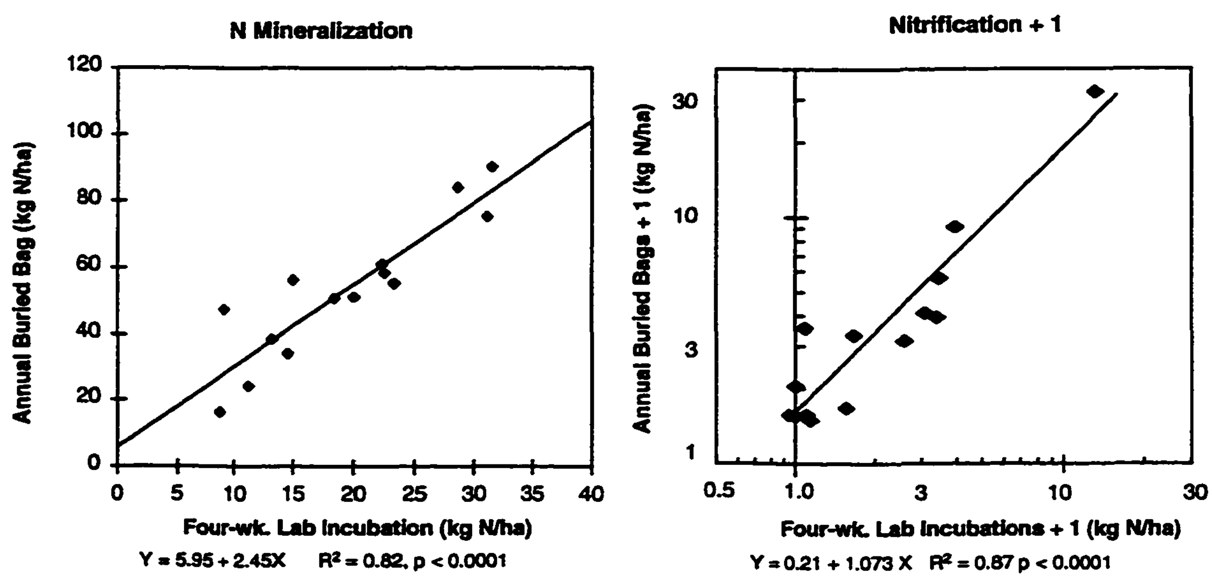

Figure 2.2: Methods comparison between 28 day laboratory and annual field measurements of a) N mineralization and b) nitrification + 1. Note the log scale for the nitrification data.

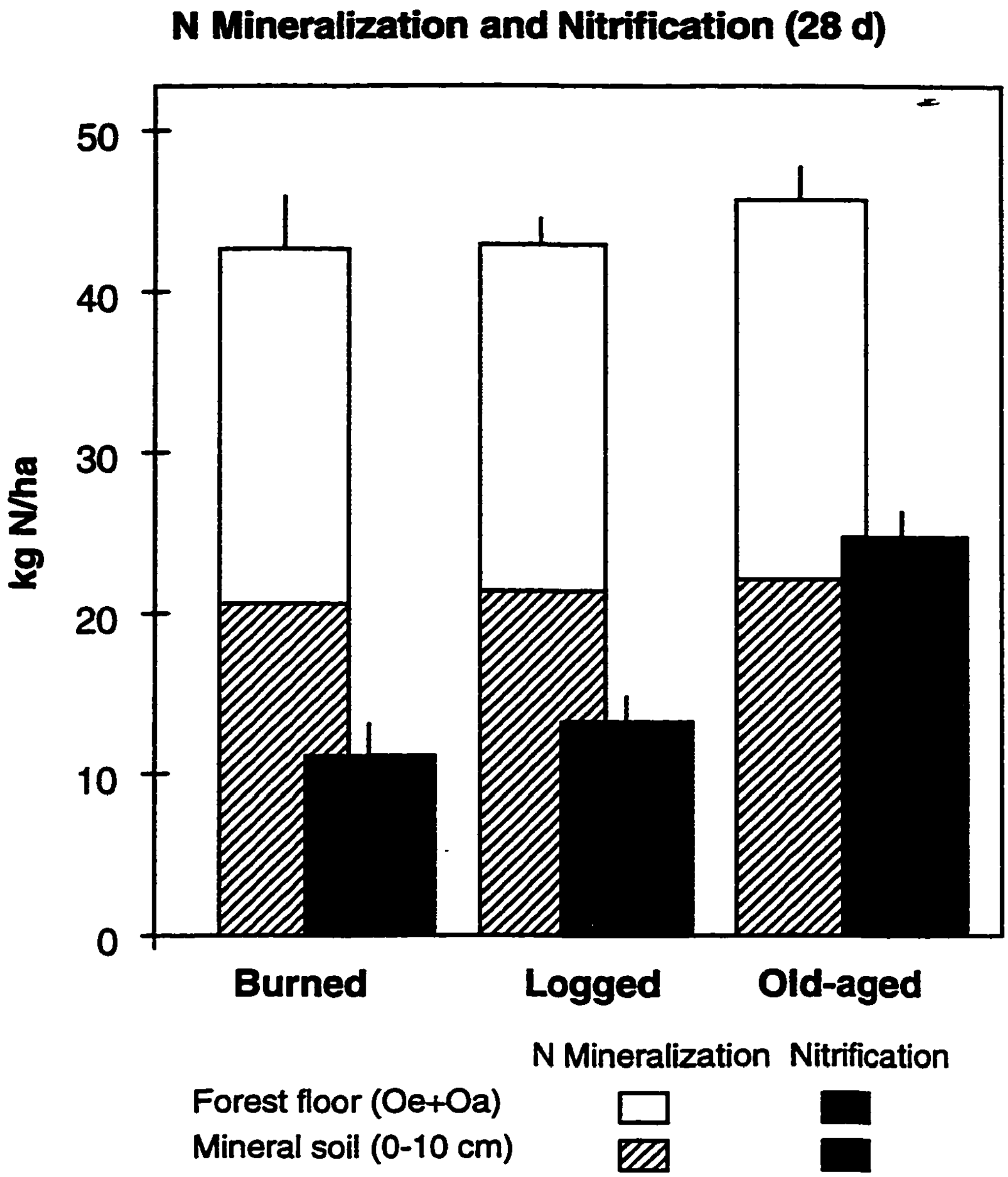

Figure 2.3: Net N mineralization and nitrification (28 d laboratory incubations) by land-use history. Bars are total mean (+ SE) values, and are divided into forest floor (upper portion) and mineral soil (lower portion). Different letters indicate significant differences by land-use history. n = 10 plots/land use.

Effects of Land-use History

Soil Processes. Nitrogen mineralization rates did not differ significantly by land-use history, whether considered on an area (kg/ha) or concentration (mg/kg) basis, by horizon, or for the total soil core (Figure 2.3, Table 2.2). However, nitrification rates did vary: nitrification rates at the old-aged sites consistently exceeded those at historically disturbed sites in both the forest floor and the mineral soil. Mean nitrification rates at the old-aged sites approximately doubled those at burned or logged sites (Figure 2.3). The fraction of total mineralized N which was nitrified (the nitrification fraction) was greater at old-aged sites relative to historically disturbed sites in all five regions. The historically logged and burned stands did not differ significantly from each other in any measured soil property (Table 2.2).

Soil Pools. The forest floors of old-aged sites had less mass, soil organic matter, C, and slightly less N than historically logged or burned sites (Figure 2.4 a, b, Table 2.2). However, the strong but not statistically significant land-use history × region interaction terms ($P = 0.06 - 0.10$) suggest that these mean differences were influenced by large differences occurring in a few regions (Table 2.2). In the mineral soil, land-use history did not affect soil organic matter, C or N content. Although land-use history differences in soil C and N content were subtle, soil C:N ratios varied strongly and consistently by land-use history in all five regions. The C:N ratios in the old-growth stands were significantly narrower than those at the historically disturbed sites, in both the forest floor and the mineral soil (Figure 2.4c, Table 2.2). Neither forest floor nor mineral soil pH varied by land-use history (Table 2.2).

Table 2.2. Mean (SE) net N mineralization, nitrification, and soil carbon, nitrogen, and C:N ratio by land use history (burned, logged, and old-aged). Different letters indicate significant differences among land use histories as determined by ANOVA blocked by region (five blocks). n = 10 for each land use type.

				Landuse (L) 2 df			Region (R) 4 df			L x R 8 df			Resid. 15 df
	Burned	Logged	Old-Aged	MS	F	P	MS	F	P	MS	F	P	MS
Net N Mineralization (kg/ha)	42.7 ± 4.3	42.8 ± 2.9	45.7 ± 2.1	30	0.5	0.63	199	3.1	0.05	136	2.1	0.10	64
Oe+Oa Forest floor	22.0 ± 1.8	21.5 ± 1.2	23.6 ± 2.1	12	1.1	0.34	83	7.7	0.001	42	3.9	0.01	11
0-10 cm Mineral soil	20.7 ± 3.5	21.4 ± 2.5	22.2 ± 2.4	5	0.1	0.90	115	2.2	0.12	94	1.8	0.15	52
Net Nitrification (kg/ha)	11.1 ± 2.9 a	13.2 ± 2.1 a	24.7 ± 2.9 b	537	6.6	**0.01**	79	1.0	0.45	45	0.5	0.80	82
Oe+Oa Forest floor	1.8 ± 0.6 a	2.5 ± 0.5 a	7.4 ± 1.3 b	92	10.6	**0.001**	14	1.6	0.22	3	0.4	0.91	9
0-10 cm Mineral soil	9.3 ± 2.7	10.8 ± 1.8	17.4 ± 2.1	185	3.7	0.05	63	1.3	0.33	43	0.9	0.57	50
Carbon (Mg/ha)	62 ± 5.8 a	62 ± 4.5 a	49 ± 3.6 b	611	6.1	**0.01**	569	5.7	0.01	287	2.9	0.04	100
Oe+Oa Forest floor	29 ± 4.8 a	25 ± 3.2 a	16 ± 2.2 b	462	6.8	**0.01**	290	4.3	0.02	157	2.3	0.08	68
0-10 cm Mineral soil	34 ± 2.9	37 ± 2.7	33 ± 1.8	42	1.5	0.25	205	7.3	0.002	55	1.9	0.13	28
Nitrogen (Mg/ha)	2.95 ± 0.2	3.05 ± 0.2	2.77 ± 0.2	0.21	1.5	0.26	1.41	10.1	< 0.001	0.38	2.7	0.04	0.14
Oe+Oa Forest floor	1.26 ± 0.2 a	1.21 ± 0.2 a	0.82 ± 0.1 b	0.58	5.8	**0.01**	0.58	5.8	0.005	0.24	2.4	0.06	0.10
0-10 cm Mineral soil	1.69 ± 0.2	1.84 ± 0.2	1.95 ± 0.1	0.17	1.4	0.28	0.74	6.0	0.004	0.13	1.0	0.45	0.12
C:N Ratio (g/g)	21.0 ± 1.1 a	20.5 ± 0.7 a	17.5 ± 0.5 b	35.9	10.8	**0.001**	18.9	5.7	0.01	6.2	1.9	0.14	3.3
Oe+Oa Forest floor	22.1 ± 0.9 a	20.9 ± 0.5 a	18.8 ± 0.5 b	27.7	10.3	**0.002**	12.7	4.7	0.01	3.4	1.3	0.34	2.7
0-10 cm Mineral soil	20.5 ± 1.2 a	20.4 ± 0.8 a	17.0 ± 0.5 b	39.4	11.1	**0.001**	23.7	6.7	0.003	8.0	2.3	0.08	3.5
pH (0.01 *M* $CaCl_2$)													
Oe+Oa Forest floor	3.5 ± 0.1	3.4 ± 0.1	3.5 ± 0.1	0.07	0.9	0.42	0.12	1.6	0.23	0.02	0.2	0.98	0.07
0-10 cm Mineral soil	3.5 ± 0.1	3.4 ± 0.1	3.6 ± 0.1	0.17	1.8	0.20	0.08	0.8	0.53	0.06	0.6	0.76	0.09

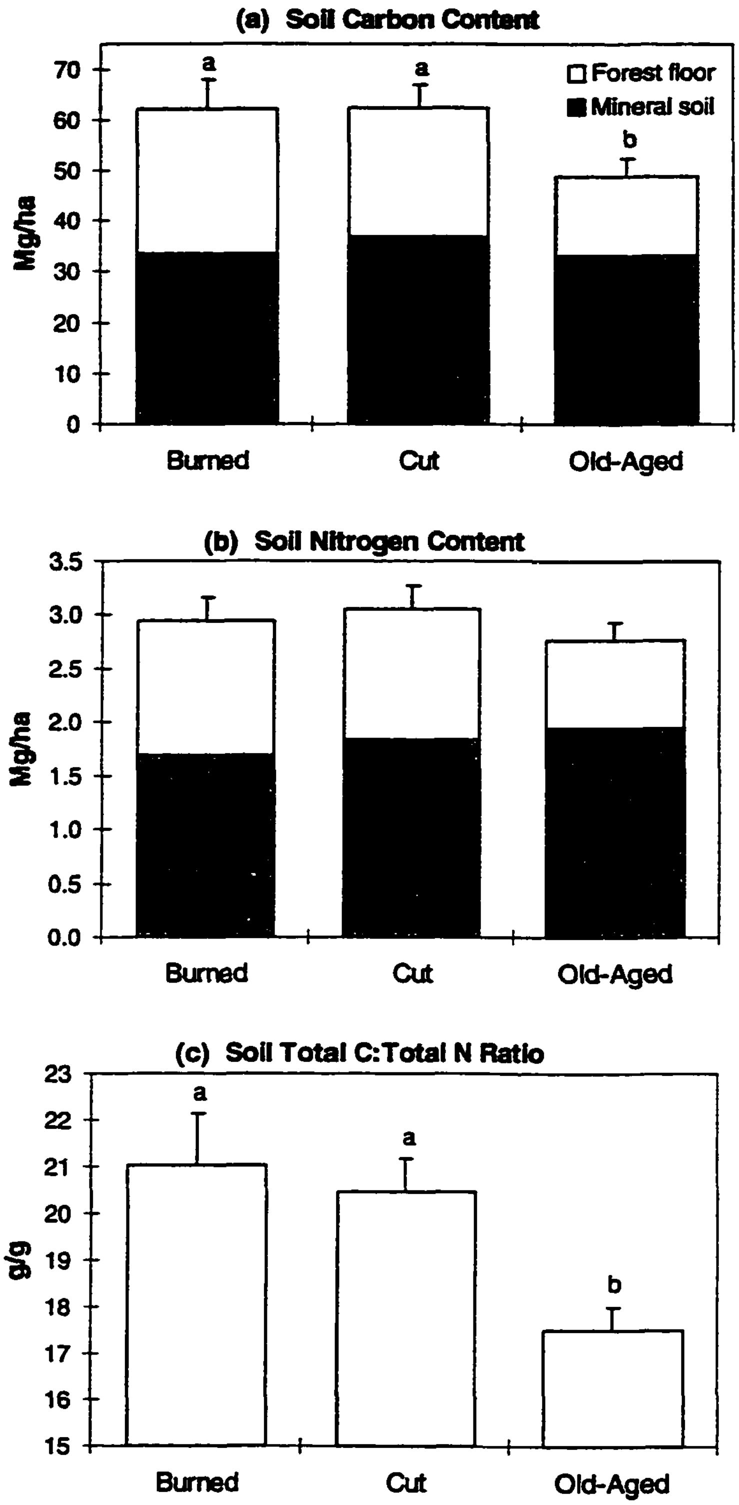

Figure 2.4: Total a) carbon and b) nitrogen content (Mg/ha) and c) C:N ratio by land-use history. Bars are mean (+SE) values, and different letters indicate significant differences among land use histories. n = 10 plots/land use.

Vegetation. Stand basal area averaged 30 m2/ha and did not vary by land-use history (df = 2, 15, $F = 0.30$, $P = 0.77$), although the historically logged and burned plots had greater numbers (755 and 615 stems/ha, respectively) of small trees while the old-aged plots had fewer (400 stems/ha), large-diameter trees (df = 2, 15, $F = 40$, $P < 0.001$). Species composition varied by land-use history in that early-successional paper birch and red maple occurred only on the historically disturbed sites (Figure 2.5). Late-successional sugar maple and American beech occurred on all three site types, but they contributed a much larger proportion of stand basal area on the old-aged sites. Conifers contributed up to 30% of the basal area of a few plots, but were minor components of most.

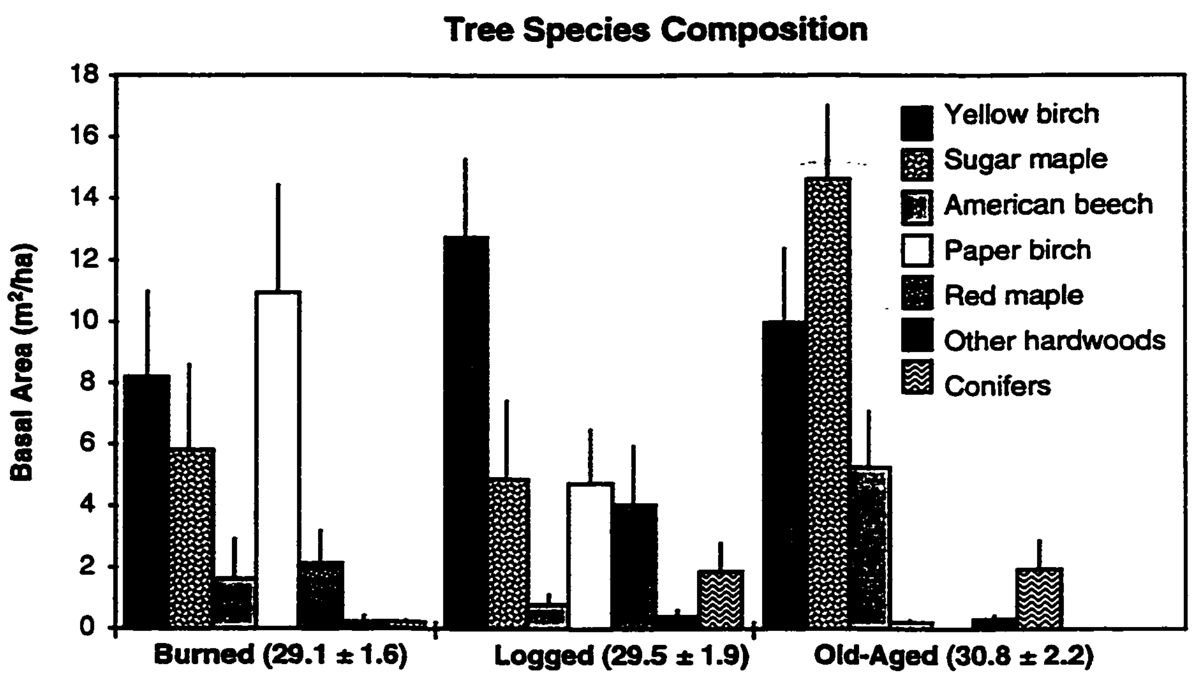

Figure 2.5: Average (+ SE) basal area (m^2/ha) by tree species for different land use types, for all trees 9.5 cm dbh or greater. Average total (SE) basal area is indicated for each land-use history.

Foliar Chemistry. Land-use history did not affect plot-averaged foliar %N, %lignin, or lignin:N ratio (Table 2.3), nor were there any differences in foliar chemistry by land use within individual species. Foliar %N averaged 2.3% for all three land-use histories. American beech, and yellow and paper birch averaged 2.5% N, while sugar and red maple averaged 2.1% N (Table 3). Of the hardwoods, sugar maple had the lowest lignin concentration (18.2%), while beech had the highest (23.9%).

Table 2.3. Mean (SD) foliar chemistry by land-use history and tree species. n indicates number of plots represented.

	n	Foliar N (%)	Foliar Lignin (%)	Foliar Lignin:N
Land Use				
Burned	10	2.33 ± 0.15	20.0 ± 1.5	8.61 ± 0.55
Logged	10	2.28 ± 0.14	21.1 ± 1.3	9.30 ± 0.73
Old-aged	10	2.31 ± 0.26	21.0 ± 1.4	9.21 ± 1.29
	$P =$	0.91	0.09	0.11
Tree Species				
Yellow Birch	24	2.53 ± 0.21	21.9 ± 1.2	8.7 ± 0.9
Sugar Maple	17	2.08 ± 0.27	18.2 ± 1.1	8.9 ± 1.2
Paper Birch	15	2.48 ± 0.15	19.6 ± 1.0	7.9 ± 0.6
Red Maple	11	2.11 ± 0.22	19.8 ± 1.2	9.5 ± 0.7
Amer. Beech	12	2.48 ± 0.18	23.9 ± 1.2	9.7 ± 0.9
Red Spruce	8	1.14 ± 0.11	26.0 ± 1.0	22.9 ± 2.6

Links Between Stream Nitrate and Soil Nitrification

Annual average stream NO_3^--N concentrations correlated remarkably well with estimates of soil nitrification, even though plots covered very small portions of stream watersheds (Figure 2.6). The old-growth sites with high nitrification rates had the highest stream NO_3^--N concentrations, while the historically disturbed sites had much lower nitrification rates and stream NO_3^--N concentrations. Nitrification rates varied greatly within both the Gibbs Brook old-aged site and the Zealand Valley burned site, suggesting patchy nitrification in both areas. Within the Gibbs Brook site, the plot with the low nitrification rates typified the mixed hardwood-conifer forest of the three large watersheds yielding very low NO_3^--N concentrations, while the plot with high nitrification rates typified the sugar maple / yellow birch composition of the small watershed yielding relatively high NO_3^--N concentrations (Figure 2.6).

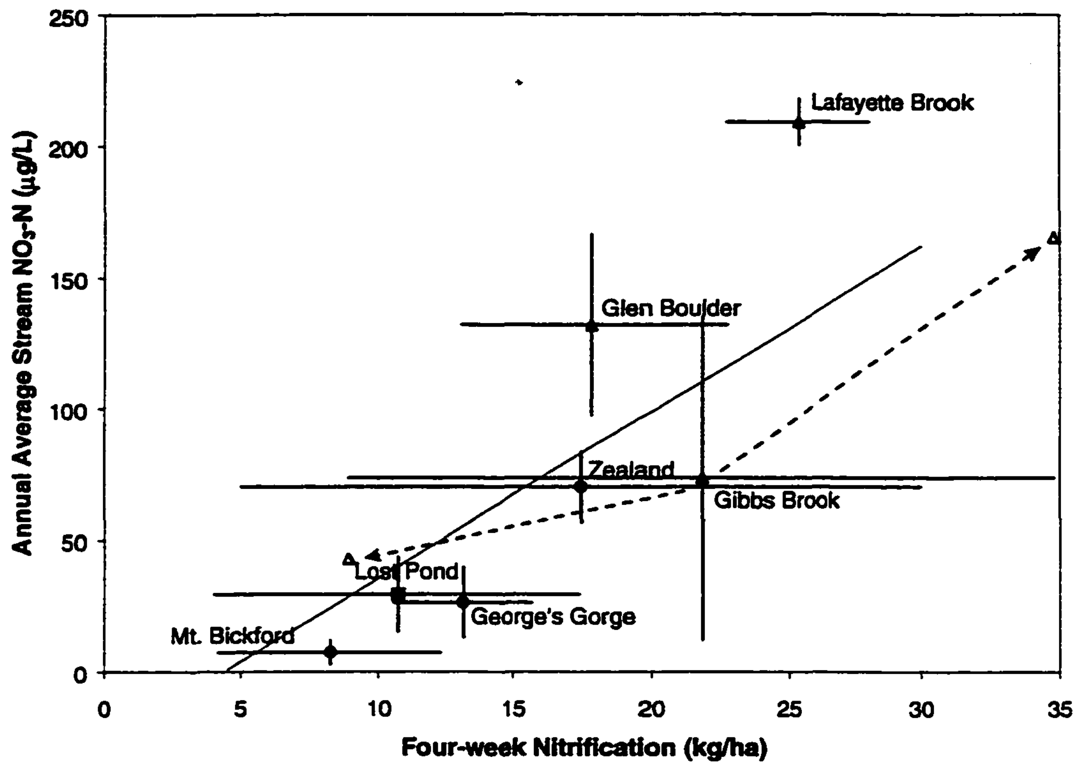

Figure 2.6: Nitrate concentrations in streams and nitrification in soils. Nitrification data are the mean (± range) of two plots per site, and stream data are averages (± SD) of annual nitrate concentration in 2-4 streams per site. Dotted lines match the two Gibbs Brook plots with related streams. Regressing the means: $Y = 6.3X - 27.6$, $R^2 = 0.73$.

Controls on Soil Nitrification

Net nitrification rates (kg/ha) correlated negatively with tree density, forest floor organic matter, total soil C, and C:N ratio, and positively with net mineralization, mineral soil pH, and sugar maple basal area and sugar maple foliar % N (Table 2.4, Figure 2.7). Sugar maple was the only species whose abundance or foliar chemistry correlated with any measured soil property. Analysis of covariance was used to determine whether the high nitrification fractions at the old-aged plots were due to the increased abundance of sugar maple or to some other aspect of land-use history. Results were not conclusive, but suggested that both land-use history (df = 2, 24, $F = 3.1$, $P = 0.06$) and sugar maple basal area (df = 1, 24, $F = 3.8$, $P = 0.06$) affected the nitrification fraction, without a significant interaction (df = 2, 24, $F = 0.4$, $P = 0.70$).

*Table 2.4. Correlation coefficients (R) for net nitrification and N mineralization and measured soil and vegetation properties. n = 30 plots for all but sugar maple foliar %N, for which n = 17. * P < 0.05, ** P < 0.01, *** P < 0.001, with no adjustments for multiple comparisons.*

Property	Nitrification	N Mineralization
Nitrification (kg/ha)	1	
N Mineralization (kg/ha)	0.63 ***	1
pH	0.64 ***	0.32
Foliar % N	0.09	0.32
Foliar % lignin	0.02	0.20
Foliar lignin:N	-0.04	-0.11
Sugar maple foliar %N	0.56 *	0.35
Sugar maple basal area (m^2/ha)	0.52 **	0.22
Conifer basal area (m^2/ha)	0.03	-0.15
Tree density (trees/ha)	-0.64 ***	-0.44 *
Forest floor mass (Mg/ha)	-0.66 ***	-0.46 *
Total C (Mg/ha)	-0.44 *	-0.20
Total N (Mg/ha)	-0.05	0.17
Total C:Total N	-0.77 ***	-0.62 ***

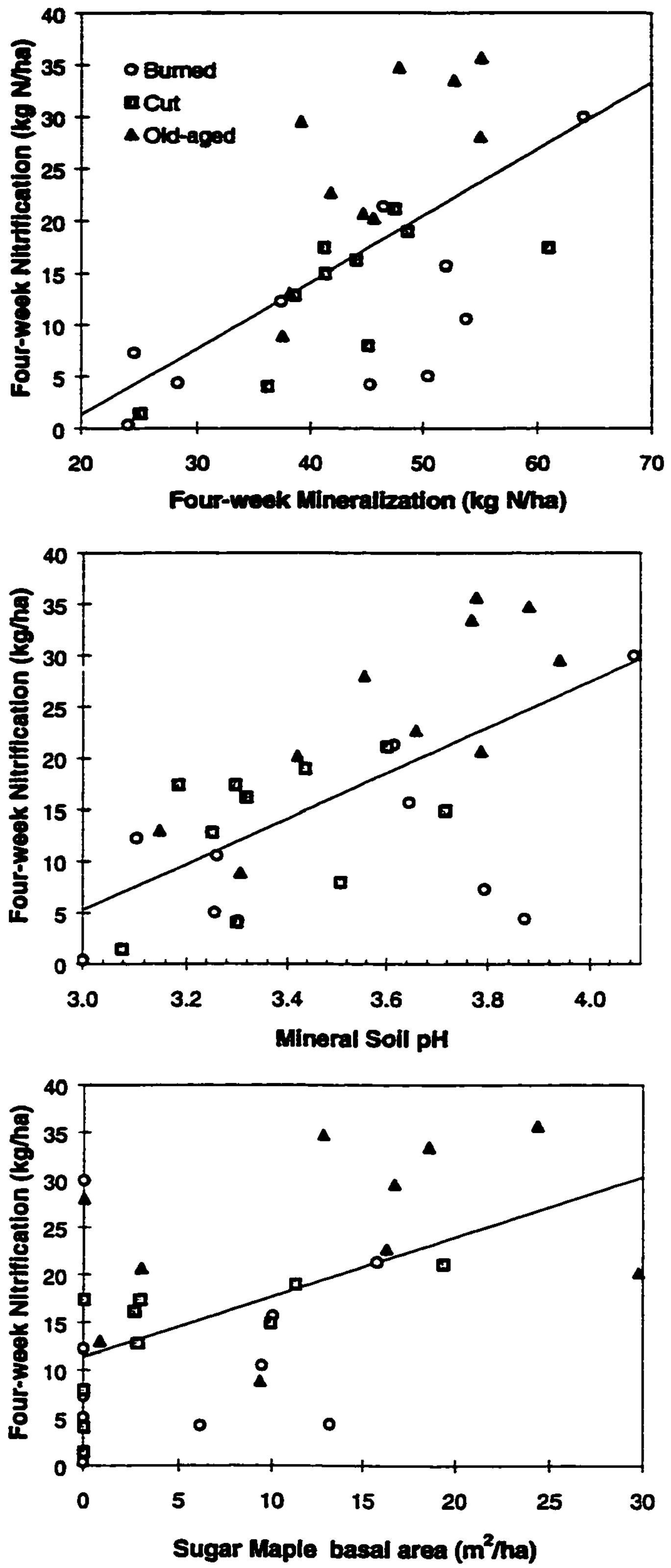

Figure 2.7: Four-week nitrification rates (kg/ha) as a function of with a) N mineralization, b) mineral soil pH, and c) sugar maple basal area. o = historically burned plots, □= logged plots, and Δ = old-aged plots.

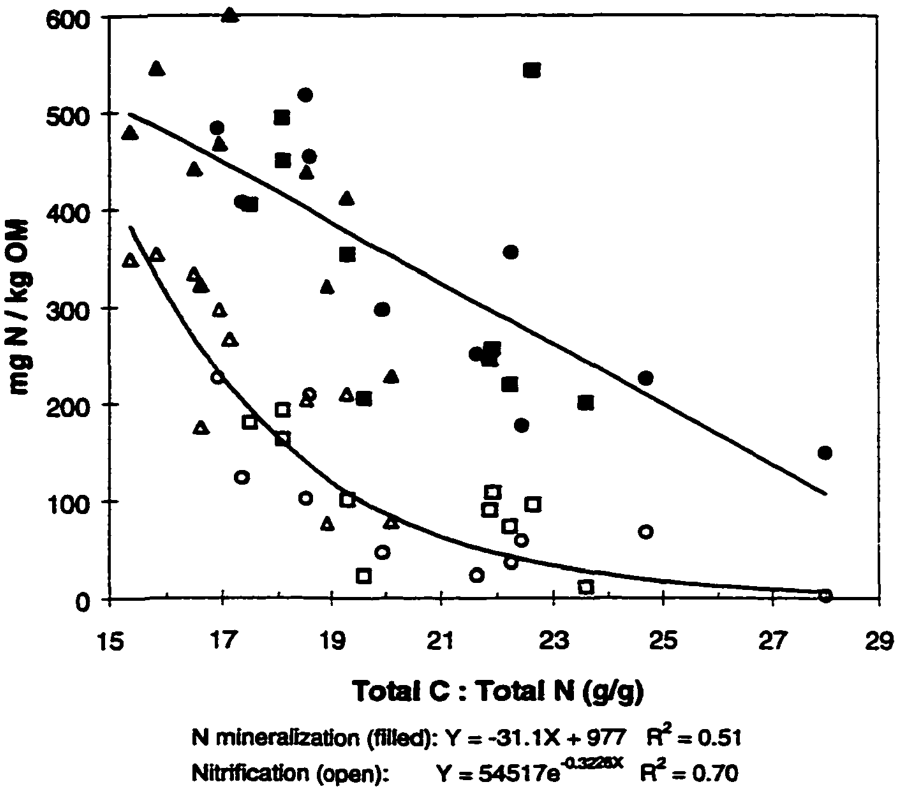

Figure 2.8: Four-week N mineralization (filled symbols) and nitrification (open symbols) as a function of soil C:N ratio decrease. o = *historically burned plots,* □= *logged plots, and* Δ = *old-aged plots.*

Multiple regression analysis using stepwise elimination of the variables in Table 2.4 retained only N mineralization, soil pH, and C:N ratio as predictors of nitrification. This model explained only slightly more variability in soil nitrification (adj. $R^2 = 0.64$) than soil C:N ratio alone ($R^2 = 0.59$). Soil N mineralization also correlated with soil C:N ratio, but the relationship was not as strong nor as steep as that between nitrification and soil C:N (Figure 2.8, $R^2 = 0.38$). Although plot-averaged foliar %N did not directly correlate with any measured soil property, it did improve prediction of N mineralization in a multiple regression model including total soil C, total N, and foliar %N (Adj. R^2 = 0.49). Relationships between N mineralization and soil C:N were particularly strong when considered per unit organic matter; nitrification appears to increase exponentially below a C:N ratio of 20 (Figure 2.8).

Discussion

Mean stand basal area, N mineralization, and soil organic matter, C, and N pools have recovered from the C and N losses incurred 80-110 years ago through logging and fire. I expected that recovery from fire would take longer than from clearcutting, yet the historically logged and burned sites had nearly identical N cycling rates and soil C and N pools. Either historical fires were less intense than presumed, or both logged and burned sites have had sufficient time to recover. However, despite this apparent recovery, nitrification rates and nitrate leaching at the historically disturbed sites remain low relative to the old-aged sites. What factors explain the differences in nitrification and nitrate leaching between the historically disturbed and the old-growth sites?

Controls on Nitrification Rates

Nitrification rates have previously been shown to relate to tree species, soil pH, ammonium supply, foliar lignin:N, and soil C:N ratio (e.g., Vitousek et al. 1982, Robertson et al. 1982, Pastor et al. 1984, McNulty et al. 1991, van Miegroet et al. 1992, Finzi et al. 1998a, Gundersen et al. 1998a). The strong link between foliar chemistry (%N or lignin:N) and ecosystem N status observed elsewhere (McNulty et al. 1991, Gundersen et al. 1998, Ollinger et al., submitted) was not observed here. The lack of correlation could be due to the small range of plot foliar chemistries observed, to inappropriate weighting of different species' chemistries (by basal area rather than relative litter production), or to differences in canopy N content rather than foliar N concentration. Of the site factors that did correlate with soil nitrification (Table 2.4), only tree species composition and soil C:N ratio varied by land-use history and could possibly explain the systematically elevated nitrification rates at the old-aged sites.

Sugar Maple. Sugar maple is a shade-tolerant, late-successional species, and nitrification rates under sugar maple frequently exceed those under other northern hardwood species (Pastor et al. 1984, Finzi et al. 1998a, Lovett and Reuth, in press). However, it is difficult to discern whether this effect is due to a particular property of sugar maple itself or simply to an underlying gradient in soil pH, texture, or mineralogy (Pastor et al. 1984, van Breeman et al. 1997). Sugar maple foliage does not have particularly high N concentrations (Table 2.3), but its low lignin content leads to low litter lignin:N ratios (Melillo et al. 1982), and its reinforcement of site calcium status through leaf litter (Finzi et al. 1998b) may cause a pH-induced increase in soil nitrification rates.

Were the high rates of nitrification observed at the old-aged sites caused by the increased abundance of sugar maple in later succession? Analysis of covariance suggested that old-aged sites continued to nitrify more than expected based on sugar maple basal area alone. The analysis indicated that the differences in sugar maple abundance could explain approximately 3 kg/ha of the observed 11-13 kg/ha N difference in nitrification rates between the disturbed and the old-aged plots. However, results from this analysis were tentative, and used plot-averaged nitrification data rather than data collected under individual trees. Species effects may have been more prominent if data were collected in this manner. The current data suggest that some additional property of the old-aged sites must account for most of the increase in nitrification rates at the old-aged sites.

Soil C:N Ratio. Total C : total N ratio was the strongest predictor of nitrification rates, regardless of land-use history (Table 2.4). Both nitrification and N mineralization increased as soil C:N decreased, although nitrification increased more steeply, and possibly nonlinearly (Figure 2.8). Other researchers have found that nitrification relates more strongly to forest floor than to mineral soil C:N ratio (Emmett et al. 1998, Gundersen et al. 1998b, Dise et al. 1998a, b). However, most of the nitrification observed in the present study and elsewhere in New England (Federer et al. 1983) occurred in the mineral soil (Figure 2.3) and nitrification rates correlated more strongly with total C : total N ratio than with C:N ratio of the forest floor alone.

The concept of critical C:N thresholds for N mineralization and nitrification was recognized long ago (reviewed in Harmsen and van Schreven 1955), but has received renewed attention in the current discussion of N saturation. Emmett et al. (1998) suggested a critical forest floor C:N ratio of 24 for the onset of nitrate leaching from European conifer plantations, a value consistent with onset of nitrification observed by Ollinger et al. (submitted) for a range of species at the Bartlett Experimental Forest, and slightly higher than the total C : total N threshold of ~ 20 indicated here (Figure 2.8).

Soil C:N ratios integrate site history of both C and N accumulation, and critical C:N thresholds should be reached fastest in areas where soil N accumulates faster than soil C. Soil N accumulation derives from N in plant litter as well as N inputs from fixation or deposition, while soil C accumulation is controlled by the balance between decomposition and inputs of fine and woody litter. Rates and quality of soil C and N inputs may change over succession as stands aggrade and change allocation patterns (Vitousek et al. 1988) or shift species composition (Van Cleve et al. 1991).

In the present study, nitrification rates correlated with soil C:N ratio and carbon content, but were not affected by soil N content alone (Table 2.4). The large differences in total C:N ratio appeared most related to differences in the C content of the forest floor. Forest floors in the old-aged stands contained less C than those of the historically disturbed stands, but this trend may have been driven by site-specific factors in a few regions. The differences in forest floor C could be due to successional patterns evolving over time, or to local site differences that were artifacts of our space-for-time (and disturbance history) substitutions. The observed trend differs from the predicted asymptotic rise in forest floor mass to 60-80 Mg/ha predicted by Covington (1981) and Federer (1984); yet current work (Yanai et al., in prep) suggests that changes in forest floor mass over time can be quite varied and site-specific. If real, the smaller forest floors at the old-aged sites could be caused by declining net primary production and organic matter inputs in the old-aged stands, or by a positive feedback between enhanced decomposition and lower C:N ratios in the old-aged stands. Reduced demand and competition for N by the old-aged forests may have allowed mineralized N to accumulate in the soil organic matter and decreased the soil C:N.

McNulty et al. (1991) and Gunderson et al. (1998a) also reported that forest floor mass decreased with C:N ratio. Forest floor organic matter and N content of the old-aged stands measured here were nearly identical to those of the Integrated Forest Study's Turkey Lakes site, an old-aged northern hardwood site in Ontario (Johnson and Lindberg 1992). The soil organic matter and N content of the historically disturbed sites strongly resembled those measured at the Hubbard Brook Experimental Forest (Johnson 1995), a nearby site that experienced heavy logging approximately 80 years ago.

Nitrification, Succession, and N Saturation

This study contributes to the long-running and often contradictory discussion of the role of succession in regulating N cycling and NO_3^- loss (e.g., Odum 1969, Rice and Pancholy 1972, Vitousek and Reiners 1975, Robertson and Vitousek 1981, Vitousek et al. 1989, Hedin et al. 1995). Vitousek and Reiners (1975) proposed that retention of limiting nutrients should decrease over successional time, and they demonstrated that New Hampshire streams draining old-aged stands had higher NO_3^- losses than stands logged approximately 30 years previously. The theory asserts that old-aged forests, with presumably low rates of net ecosystem production, require few of the nutrients made available by atmospheric deposition or weathering. With decreased forest demand for N, nitrification and nitrate leaching should increase over successional time. This pattern of elevated nitrification rates or nitrate losses in old-aged stands relative to successional forests has been demonstrated elsewhere in the southeastern (Silsbee and Larson 1982, Sasser and Binkley 1989, Flum and Nodvin 1995) and the northeastern (Leak and Martin 1975, Martin 1979, Robertson and Vitousek 1981, this study) U.S. However, old-aged forests on the west coast of North (Sollins et al. 1980) and South (Hedin et al. 1995) America often have quite low rates of nitrification or nitrate loss. Short-term differences in species' allocation of C and N and long-term differences in N deposition may explain many of these differences among sites.

Successional sequences that lead from old-field grasses to forests, or from hardwoods to conifers frequently report decreasing nitrification rates and increasing soil C:N ratios in the first several decades of succession (Van Cleve et al. 1991, Thorne and Hamburg 1985, Zak et al. 1990, Compton et al. 1998). These trends in soil C:N parallel the expected increase in litter C:N as woody plants replace grasses or conifers succeed hardwoods. Conifers tend to have low N concentrations in both fine and woody litter (Gosz 1981), and site or successional factors that increase conifer abundance can increase the C:N ratio of soil organic matter (Van Cleve et al. 1991). In this set of studies, the rate of soil C accumulation has outpaced the rate of N accumulation at least temporarily, and decreased nitrification rates corresponded with the increased soil C:N ratios.

Elevated rates of N deposition can enrich soil N, leading to narrower soil C:N ratios over time (McNulty et al. 1991, van Miegroet et al. 1992, Tietema and Beier 1995). Areas that receive very little N deposition may require an extremely long time to receive enough N to narrow soil C:N ratios through accrual over succession; several hundred years may not suffice. Old-growth angiosperm *(Nothofagus)* forests in western Chile receive very little N deposition, have soil with relatively high C:N ratio (33), and lose extraordinarily small amounts of nitrate in streams (Hedin et al. 1995, Pérez et al. 1998). In contrast, the old-aged northern hardwood forests in this study and at Turkey Lakes receive elevated rates of N deposition, have low C:N ratios (15-17.5), and leak nitrate (Foster et al. 1989, Johnson and Lindberg 1992, Mitchell et al. 1992). Conifer forests receiving chronic N deposition also exhibit elevated nitrification or nitrate loss (Vitousek and Reiners 1975, Sasser and Binkley 1989, Friedland et al. 1991, Emmett et al. 1998).

I conclude that disturbances 80 to 110 years ago have had long-term effects on N cycling by allowing these aggrading forests to continue to retain deposited N in both soils and aboveground vegetation. Tree basal area and soil organic matter pools have recovered in the historically disturbed stands, yet nitrate production and losses to streamwater remain low relative to old-aged stands. The elevated nitrification rates in the old-aged stands correspond with narrow soil C:N ratios, likely resulting from chronic N deposition combined with age-induced reductions in organic matter inputs or enhanced decomposition. The combination of old age and chronic N deposition makes eastern old growth forests particularly vulnerable to N saturation.

CHAPTER III

CHANGES IN WHITE MOUNTAIN STREAM CHEMISTRY OVER TWO DECADES

Abstract

Twenty-eight streams on Mt. Moosilauke and Mt. Washington, NH, sampled throughout 1973-4 (Vitousek 1977) were located and sampled seasonally during 1996-7. The streams provided the opportunity to evaluate forest response to 23 years of forest aggradation and to chronic N and declining SO_4^{2-} and base cation deposition. On Mt. Moosilauke, estimated annual average NO_3^- concentrations declined by 68% (22 μeq/L); Ca^{2+}, by 28% (18 μeq/L); Mg^{2+}, by 26% (10 μeq/L); and SO_4^{2-}, by at least 22% (20 μeq/L). Stream pH and calculated acid neutralizing capacity (ANC) increased, particularly at previously acidic spruce-fir watersheds. Every stream had lower NO_3^- concentrations in 1996-7 than in 1973-4, but spatial patterns among streams persisted: streams draining old-aged stands maintained higher NO_3^- concentrations than those draining successional stands. These changes in ion concentration are consistent with the long-term record at the Hubbard Brook Experimental Forest, NH over 1973-1994. Declines in surface water SO_4^{2-} and base cation concentrations are consistent with decreasing SO_4^{2-} and base cation deposition since the 1970s, yet N deposition has changed little. Soil frost and insect outbreaks cannot fully explain the sharp NO_3^- decline. Climate variability and its effects on biotic N retention may be responsible for the synchronous decreases in NO_3^- concentrations across all streams, overriding expected increases due to chronic N deposition or forest aging. Forest successional status appears

to influence spatial patterns of NO_3^- leaching, but it seems that climate controls temporal patterns on the scale of months to decades.

Introduction

Long-term records of stream chemistry allow the study of ecosystem response to changing or chronic sulfate, nitrogen, and base cation deposition. Sulfur dioxide emissions from the northeastern and mid-western U.S. have declined considerably since the Clean Air Act of 1970, and precipitation SO_4^{2-} concentrations have declined in response (Hedin et al. 1987, Dillon et al. 1988, Butler and Likens 1991, Stoddard et al. 1998b). Precipitation pHs generally increased in response to the declines in SO_4^{2-} (Butler and Likens 1991), although not as much as anticipated due to concurrent declines in base cation concentrations (Hedin et al. 1987, 1994). Nitrogen emissions received little attention during early research and legislation on acid deposition, and NO_3^- concentrations in northeastern U.S. precipitation have not changed significantly over the past three decades (Dillon et al. 1988, Butler and Likens 1991, Stoddard et al. 1998b, Driscoll et al. submitted).

Surface water SO_4^{2-} concentrations have declined almost uniformly across the northeastern U.S. since the early 1980s (Stoddard et al. 1998a; b). Yet, surface water acid neutralizing capacities (ANC) and pHs have not rebounded as anticipated due to compensating declines in base cation concentrations (Driscoll et al. 1989, 1995, Stoddard and Kellogg 1993, Likens et al. 1996, Stoddard et al. 1998a, b) or increases in stream NO_3^- concentrations (Stoddard 1991, Murdoch and Stoddard 1992, 1993, Driscoll and Van Dreason 1993). Research on watershed acidification has thus shifted focus from the effects of SO_4^{2-} deposition only, to the combined effects of attenuated SO_4^{2-} deposition, decreased deposition of base cations, and chronic inputs of nitrogen.

Chronic nitrogen deposition may lead to N saturation, which is N availability in excess of plant and microbial demand, accompanied by elevated nitrification and NO_3^- leaching (Aber et al. 1989, Stoddard 1994). Nitrogen deposition has led to N saturated forests in the eastern and western U.S. (reviewed in Fenn et al. 1998) and in western Europe (Emmett et al. 1998). Both chronic N deposition and forest succession may decrease forest N retention. Vitousek and Reiners (1975) used stream data from Mt. Moosilauke, New Hampshire, to support the hypothesis that system retention of limiting nutrients decreases with stand age. Streams draining old-aged stands on Mt. Moosilauke had much higher concentrations of NO_3^- than did streams draining young successional stands.

In 1996-7, I located and sampled these and other streams on Mt. Moosilauke and Mt. Washington, New Hampshire which were all sampled throughout 1973-4 (Vitousek 1977). The 1973-4 data provide a baseline of forest and stream response during a period of heavy atmospheric deposition. They also provide an opportunity to evaluate whether patterns of N loss observed in 1973-4 persisted, and whether N losses had increased after 23 years of forest maturation and chronic N deposition. Observations of stream chemistry over two isolated years cannot substitute for long-term data; yet, they can indicate relative changes across different systems, and comparison with the continuous long-term record at the nearby Hubbard Brook Experimental Forest (Driscoll et al. 1989, Likens and Bormann 1995) allows some general inferences of long-term change across the region.

Methods

Study Sites

Mt. Washington, the highest peak in the northeastern U.S. (1917 m), rises from the north-central portion of the White Mountain National Forest, New Hampshire (44° 16'N, 71° 18'W), while Mt. Moosilauke (1464 m) lies on the southwestern edge of the White Mountain region (44° 1'N, 71° 50'W; Figure 3.1). The Hubbard Brook Experimental Forest is 12 km southeast of Mt. Moosilauke's summit. On both mountains, vegetation grades with elevation from northern hardwood (< 750 m) to spruce-fir (750-1200 m), subalpine balsam fir (1200-1400 m), and alpine tundra (> 1400) (Leak and Graber 1974, Reiners and Lang 1979). The composition, structure, and nutrient cycling properties of northern hardwood (e.g., Bormann and Likens 1979, Likens and Bormann 1995), spruce-fir (Foster and Reiners 1983, Huntington et al. 1990, Friedland et al. 1991) and subalpine balsam fir (Reiners and Lang 1979, Lang et al. 1981, Sprugel 1984) forests have been discussed elsewhere.

The bedrock of both mountains was previously classified as Littleton formation mica schist (Billings 1956). Mt. Washington retains this designation, but the geology of Mt. Moosilauke is now classified as quartz-feldspar-biotite metagraywacke (dark, metamorphosed sandstone and fine particles; Hatch and Moench 1984). Ammonoosuc Volcanics occur off Mt. Moosilauke's western slope, containing large amounts of calcium-rich hornblende and plagioclase feldspar relative to the base-poor metamorphic formations (Bailey and Hornbeck 1992). Glacial movement from northwest to southeast transported and deposited till such that locally-derived soils often resemble the bedrock mineralogy up to 32 km "upglacier" (Bailey and Hornbeck 1992; Hornbeck et al. 1997). This glacial transport 14,000 years ago may have enriched soils on the western flank of Mt. Moosilauke with calcium-rich minerals.

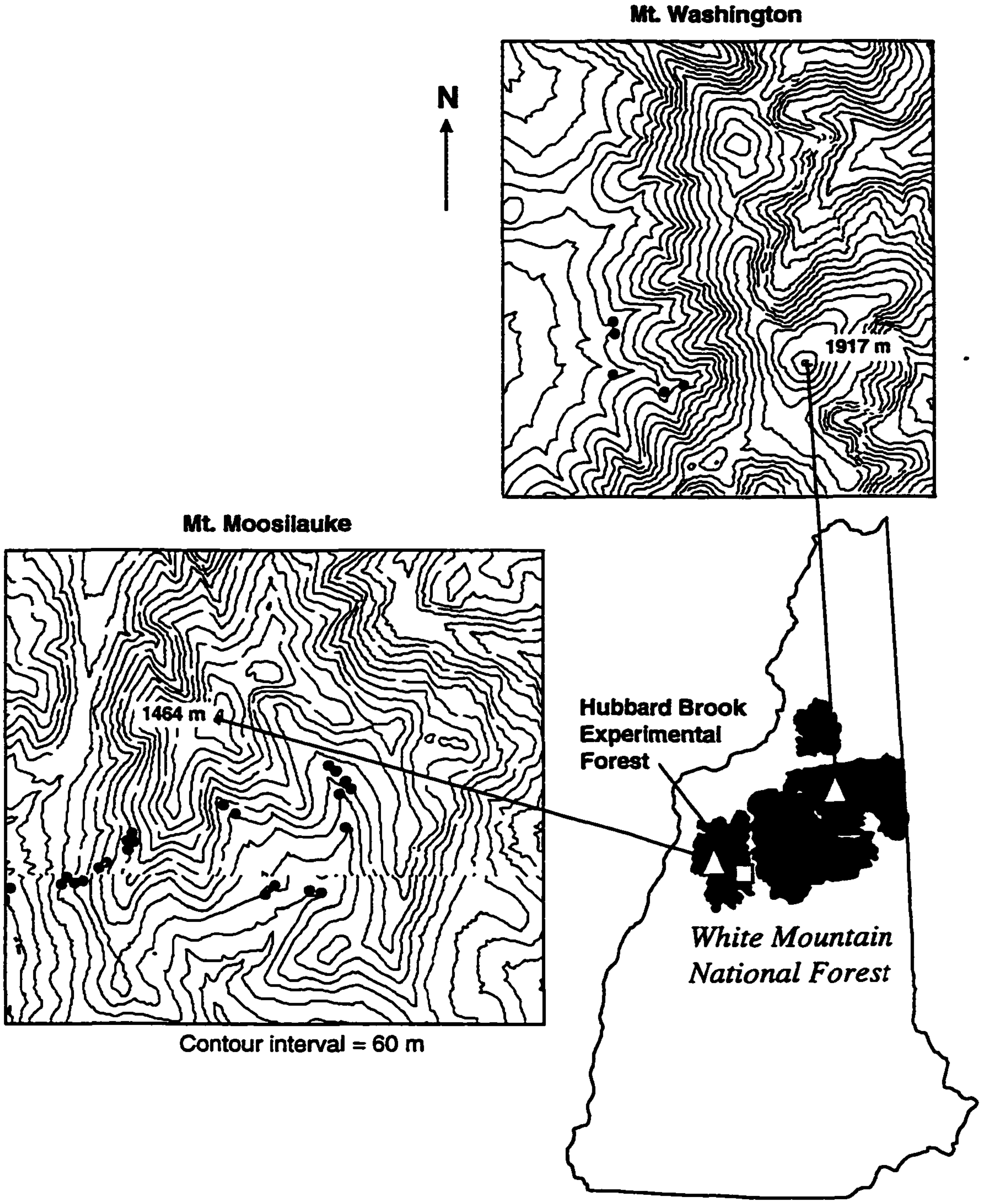

Figure 3.1: Study area and stream sampling locations on Mt. Moosilauke and Mt. Washington, White Mountain National Forest, New Hampshire.

The climate is cool and temperate, and 1-2 m snowpacks generally accrue during winter. At Hubbard Brook, annual precipitation averages 1300 mm, distributed evenly throughout the year, and monthly mean temperatures range from -8.7 °C in January to 18.8 °C in July (Federer et al. 1990). Streamflows generally peak with April snowmelt. Due to the cool, humid climate and stony, base-poor parent material, haplorthods developed across most of the region, grading to cryorthods and cryofolists at higher elevations (Huntington et al. 1990).

Forest history on both mountains is marked by wide-spread clearcutting near the turn of the century. On Mt. Washington, accessible spruce stands were cut heavily around 1910, while the subalpine forests on the steep upper slopes were left uncut (USDA Forest Service records, Laconia, NH). On the southwestern side of Mt. Moosilauke, forests on the lower slopes were cut heavily prior to 1901. Red spruce was cut selectively from the upper elevations, but minimally disturbed northern hardwoods and a small patch of uncut spruce remain (Brown 1958, Cogbill 1989, USDA Forest Service records Laconia, NH). On the eastern side of Moosilauke, heavy cutting between 1896 and 1923 stripped timber from all but one ravine. This ravine was salvage-logged after the 1938 hurricane (1943-1947), but stands on the ravine's upper rim were not cut (Brown 1958, Cogbill 1989). Vegetation on many of the cutover sites currently consists of a canopy of paper birch and other hardwoods with abundant spruce and fir in the subcanopy and understory.

Of the original 57 streams sampled in 1973-4 (Vitousek 1977), 16 drained primarily high-elevation alpine tundra or fir krummholz, and were not resampled. Of the remaining forested watersheds, eight could not be relocated, and five had been clearcut or thinned between 1974 and 1997 (USDA Forest Service records, Plymouth, NH). Of the remaining 28 streams, five drained the western slope of Mt. Washington, and the rest occurred on the southwestern (n = 11) and eastern (n = 12) slopes of Mt. Moosilauke (Figure 3.1). All streams were sampled along hiking trails at essentially the same locations as in 1973-4.

Sample Collection and Analysis

Vitousek (1977) describes in detail the sampling and analytical methods used in 1973-4. Differences in sampling frequency and analytical methods are identified here (Table 3.1), and discussed later. From May 1973 - Oct. 1974, the Mt. Moosilauke streams were sampled every 2-4 weeks. Data from Oct. 1973 - Sept. 1974 are included here to coincide with the Oct. - Sept. water year used in 1996-7 and in other published data (Hornbeck et al. 1997). During 1996-7, the Moosilauke streams were collected quarterly: in fall (Nov. 11 & 14, 1996), winter (Jan. 21 & 23, 1997), snowmelt (Apr. 8 & 10, 1997), and during the growing season (July 21, 1997). Some streams froze solid in winter and could not be collected. Three streams with too few collections were not included in annual analyses. The Mt. Washington streams are not easily accessed in winter, and were collected from June through Sept. in both years. Samples were collected every 2-3 weeks in 1973-4 and monthly in 1996-7.

A separate network of White Mountain streams was sampled monthly from Oct. 1996 to Sept. 1997 with collection and analytical methods identical to those in this study (Chapter 1). Data from 26 of these streams were used to test the validity of the quarterly sampling scheme used on Mt. Moosilauke.

All 1996-7 samples were collected in 250 mL high-density polyethylene bottles washed with 10% HCl and rinsed repeatedly with deionized water. At collection, bottles were rinsed with streamwater three times and then fully filled and capped. Samples were refrigerated until processing the following day. Streamwater pH was measured at room temperature within 24 hours of collection with an Orion® combination electrode. No pH data are presented for Nov. 1996 due to malfunction of the pH meter. Samples were then suction filtered through ashed (1 hour at 425 °C) Whatman GF/F glass fiber filters into sidearm flasks, and frozen in designated polyethylene scintillation vials until chemical analysis. After filtration through 0.2 μm pore size Acrodisks (Gelman Sciences), NO_3^-, SO_4^{2-}, and Cl^- were measured with a Waters ion chromatograph and a Dionex AS4A column with micromembrane chemical suppression. Base cations (Ca^{2+}, Mg^{2+}, Na^+, and K^+) were measured with direct current plasma emission spectroscopy (SpectraSpan III, ARL Direct Current Plasma, Fisons Instruments Inc., Danvers, MA).

At each collection, 3-5 blank samples were carried through all collection, filtering, storage, and analytical procedures. Mean blank concentrations of 4.2 μeq/L Cl^-, 1.5 μeq/L K^+, and 9.3 μeq/L Na^+ indicated contamination of these ions from acid washing or filtering. However, mean blank Ca^{2+}, Mg^{2+}, NO_3^-, and SO_4^{2-} concentrations were all below analytical detection limits (<1 μeq/L). Mean blank concentrations for each collection were subtracted from measured sample values.

Acid neutralizing capacity (ANC) was calculated from the chemistry available for both time periods as (μeq/L):

$$ANC = ([Ca^{2+}] + [Mg^{2+}] + [Na^+] + [K^+]) - ([NO_3^-] + [SO_4^{2-}] + [Cl^-]) \qquad (1)$$

ANC represents the ability of a surface water to absorb strong acids without increasing H^+ or Al^{3+} mobility. ANC will only increase in response to declining acid deposition if the concentration of acid anions declines faster than the concentration of base cations.

Table 3.1: Analytical methods and limits of detection (μeq/L). Vitousek (1977) reported analytical detection limits for 1973-4. In 1996-7, limits of detection were calculated as (X + 3 SD) blanks.

	1973-4		1996-7	
	Method	**LOD**	**Method**	**LOD**
Nitrate	Nitrate reduction & ammonia-specific electrode	N/A	Ion chromatography	0.4
Sulfate	Turbidimetric method	30	Ion chromatography	2.1
Chloride	Silver-specific ion electrode	3	Ion chromatography	2.7
Ammonium	Ammonia-specific electrode	1	Automated phenolate method	1.0
Calcium	Atomic Absorption	0.5	Direct current plasma	1.4
Magnesium	Atomic Absorption	0.2	Direct current plasma	0.3
Sodium	Atomic Absorption	1	Direct current plasma	1.6
Potassium	Atomic Absorption	0.5	Direct current plasma	2.6
pH	Potentiometric electrode		Potentiometric electrode	

Quality Assurance

To ensure instrument and calibration accuracy, independent commercial standards were included during all 1996-7 analyses. Measurements of independent anion and cation standards were always within 5% and 10%, respectively, of their true values. Duplicate measures of samples had coefficients of variation (CVs) of less than 2% for NO_3^- and SO_4^{2-}, less than 8% for Cl^-, generally less than 10% for Ca^{2+}, Mg^{2+}, and Na^+, and less than 15% for K^+. The higher CVs for K^+ occurred because concentrations were so low (mean = 9 μeq/L).

Daily streamflow through 1997 and monthly stream chemistry through 1992 were obtained for Hubbard Brook Watershed 6 (HB W6) from the Hubbard Brook Ecosystem Study World Wide Web site (http://www.hbrook.sr.unh.edu/data/data.htm). Federer et al. (1990) have published the streamflow data through 1988, and the stream chemistry has been published through 1992 in Likens and Bormann (1995). Hornbeck et al. (1997) have published HB W6 stream chemistry for 1992-3 and 1993-4.

The 1973-4 data were checked by comparison with stream chemistry at HB W6, and by calculating ion balances. The major concern about the reliability of the 1973-4 data involved SO_4^{2-}. Stream data for 1974 showed a large jump in early summer (5/29-

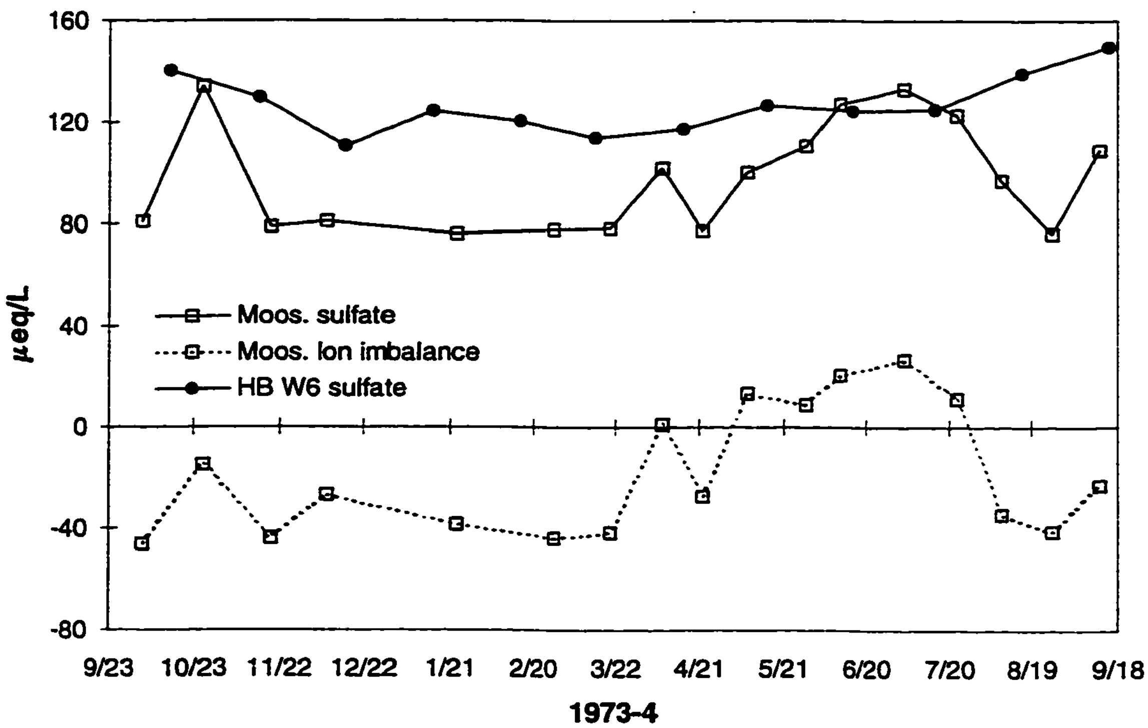

Figure 3.2: The mean 1973-4 sulfate concentrations and calculated ion imbalance vary erratically in five eastern slope hardwood streams, Mt. Moosilauke, while Hubbard Brook W6 sulfate concentrations remain relatively constant.

7/23) SO_4^{2-} concentrations while HB W6 SO_4^{2-} remained nearly constant (Figure 3.2). Measured charges did not balance during this period: in some streams with measured pH > 5.5, measured anions exceeded measured cations substantially (mean = 25-33 μeq/L) and unmeasured Al^{3+} cannot account for the discrepancy. The turbidimetric method used for SO_4^{2-} analysis in 1973-4 is relatively imprecise (Golterman 1969, Greenberg et al. 1992) but the observed concentrations should be within its range. Still, it appeared that the SO_4^{2-} results from these early summer collections were inaccurate, and they were excluded from calculations of mean annual or growing season SO_4^{2-} concentration and ANC. These removals lowered estimates of annual SO_4^{2-} concentration on Mt. Moosilauke by an average of 11 μeq/L (range 6-21 μeq/L) and estimates of growing-season SO_4^{2-} concentration on Mt. Washington by an average of 19 μeq/L (range 10-28 μeq/L). This approach led to conservative estimates of SO_4^{2-} decline and ANC increase between 1973-4 and 1996-7.

Data Summary

For the Mt. Moosilauke streams, annual (Oct. - Sept.) mean concentrations were calculated for all ions for both years. The annual means for 1973-4 summarize 18 collections for all but SO_4^{2-}, which included only 14 collections. For 1996-7, annual values were estimated as the average of the quarterly collections. For the Mt. Washington streams, growing season (Jun. - Sept.) means were calculated for both 1974 and 1997. Nitrate, the only ion that showed substantial seasonal variation, was compared across years both seasonally and annually, while the other ions were only compared annually.

Results

Quarterly Sampling: 1996-7

The accuracy of deriving annual means from quarterly (Nov., Jan., Apr., and July) samples was tested with monthly data from 26 additional White Mountain streams. Annual ion concentrations derived from averages of quarterly measurements did not differ from means derived from monthly collections (Figure 3.3a,b). For pH, only three months (Jan., Apr., and July) were compared because the Moosilauke streams were

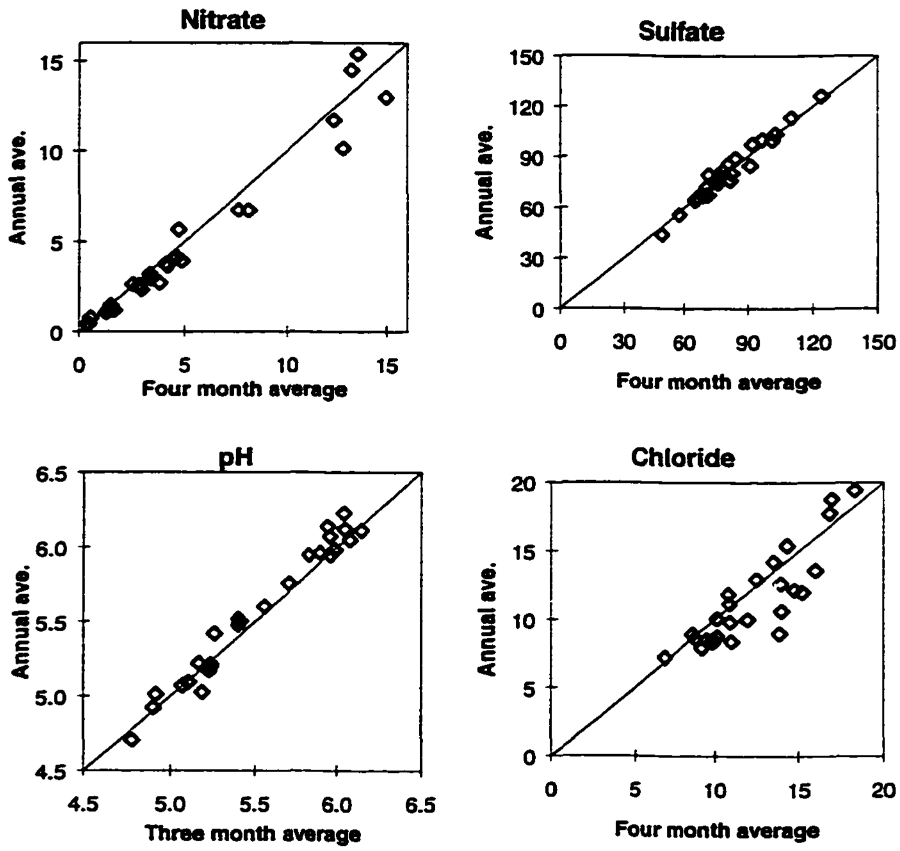

Figure 3.3a: Comparison of annual average nitrate, sulfate, and chloride concentrations (μeq/L) and pH derived from quarterly and monthly sampling schedules for 26 White Mountain streams. Lines are 1:1 lines.

missing these data for Nov. Paired-sample t-tests detected no significant differences between 4(3)-month and 12-month averages for any measured ion ($P = 0.12 - 0.71$), and mean absolute errors were less than 2.9 µeq/L. These results support the use of mean quarterly chemistry to estimate annual values for 1996-7 for the Mt. Moosilauke streams.

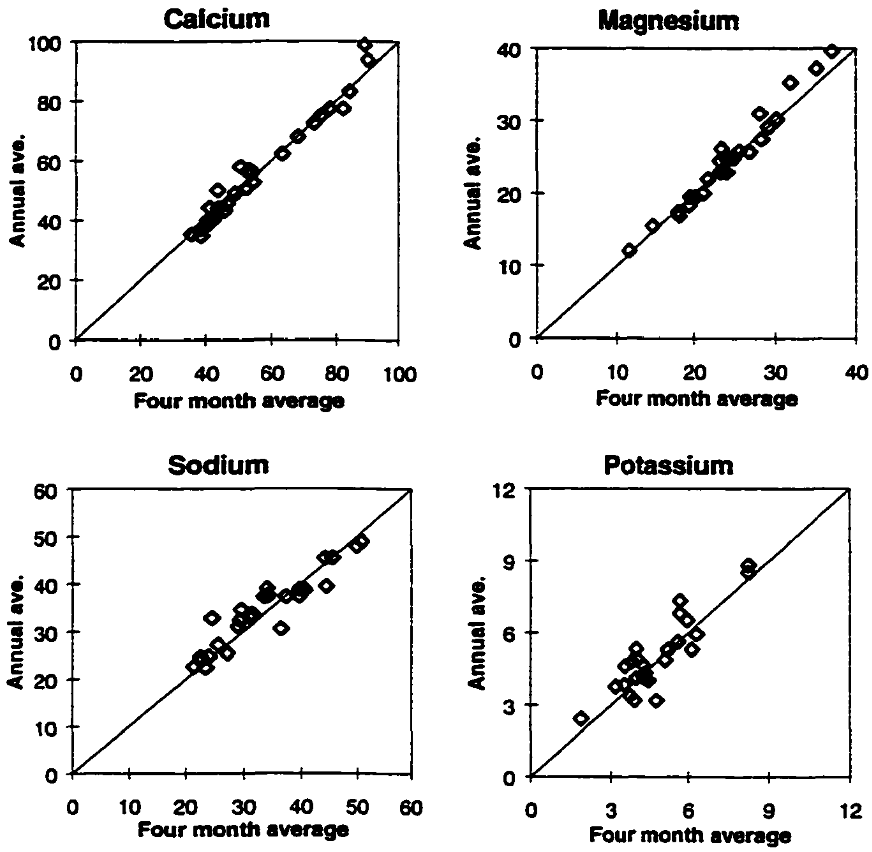

Figure 3.3b: Comparison of annual average calcium, magnesium, sodium, and potassium concentrations (µeq/L) derived from quarterly and monthly sampling schedules for 26 White Mountain streams. Lines are 1:1 lines.

Between-Year Comparisons

1973-4 and 1996-7 annual ion concentrations for 20 Mt. Moosilauke streams were compared with paired difference t-tests with Bonferroni corrections for multiple comparisons (alpha = 0.05/8 = 0.006). The greatest change in stream chemistry between 1973-4 and 1996-7 was the large decrease in NO_3^- concentration observed at all streams at all sampling periods. On Mt. Moosilauke, NO_3^- concentrations in 1996-7 averaged 22 µeq/L (68%) lower than in 1973-4; Ca^{2+}, 18 µeq/L lower (26%); and Mg^{2+}, 10 µeq/L lower (24%) (Figure 3.4a). Corrected SO_4^{2-} concentrations declined an average of 20 µeq/L (21%), mean ANC increased by 16 µeq/L, and mean Na^+ concentrations did not change.

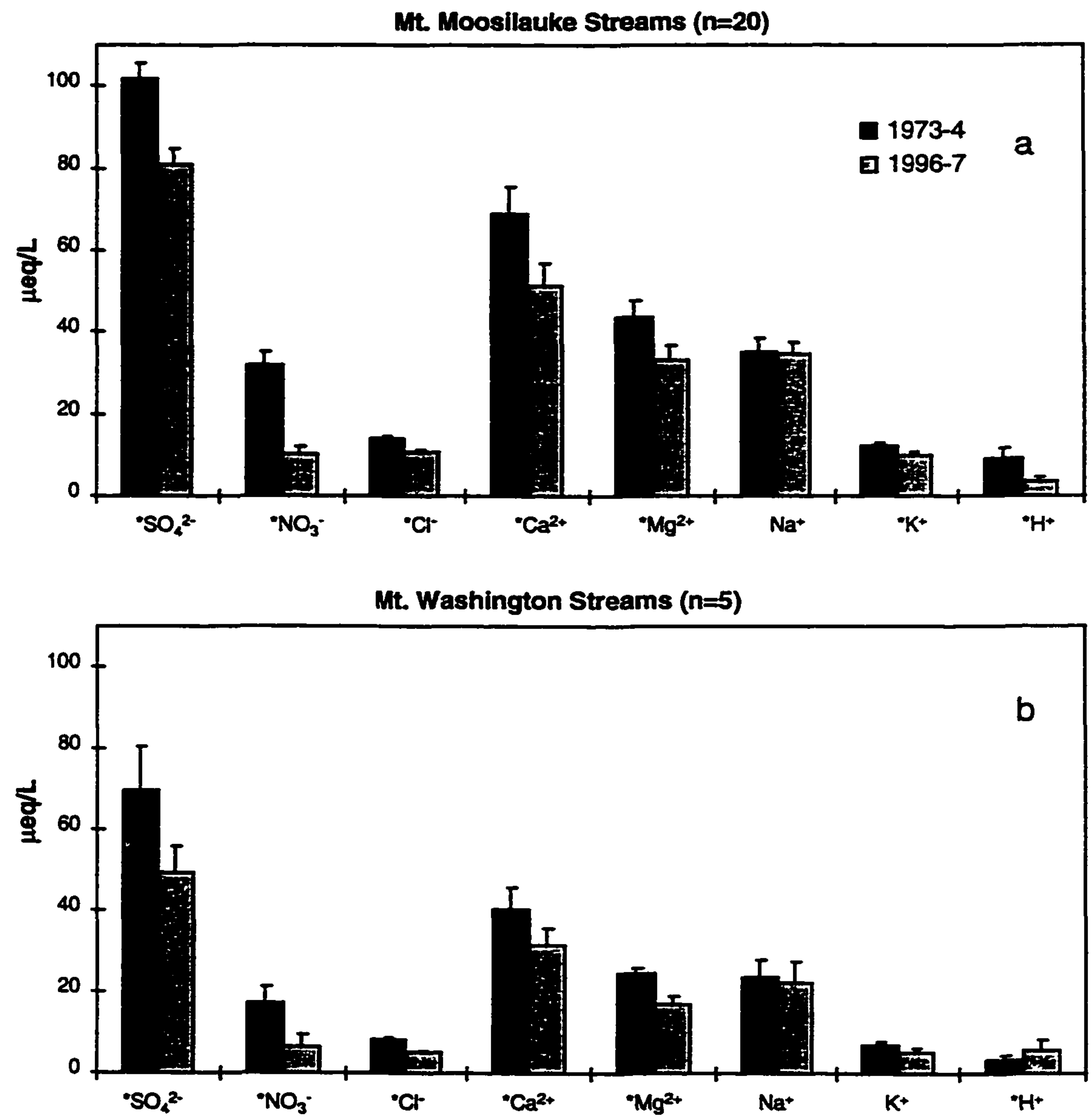

Figure 3.4: Mean (+ SE) annual ion concentrations (µeq/L) of the same streams on a) Mt. Moosilauke and b) Mt. Washington in 1973-4 and 1996-7. Standard errors represent the variability among streams, and asterisks indicate significant differences between years.

Mean pH increased significantly from 5.6 to 5.8. Declines in K^+ and Cl^- concentrations averaging 2 and 3 μeq/L were statistically significant but equivocal, since variability in sampling, analyses and annual averaging could easily have generated biases of this magnitude. Similar percent declines occurred in growing-season average ion concentrations in the Mt. Washington streams (Figure 3.4b), although all ion concentrations were generally lower on Mt. Washington. Orographic rainfall, steep, rocky, drainages, and limited evapotranspiration in high-elevation tundra likely contributed to the dilution of Mt. Washington streams.

Qualitative chemical trends among streams generally persisted across the years: streams with high concentrations of particular ions in 1973-4 also had the highest concentrations in 1996-7 (Figure 3.5). Between the two years, NO_3^-, Ca^{2+}, and Mg^{2+} concentrations decreased consistently across all resampled streams. Stream pH and ANC changed little in non-acidic streams, but generally increased in the streams that were most acidic in 1973-4. Changes in the chemistry of HB W6 between 1973-4 and 1993-4 are displayed relative to the resampled Mt. Moosilauke and Mt. Washington streams (Figure 3.5).

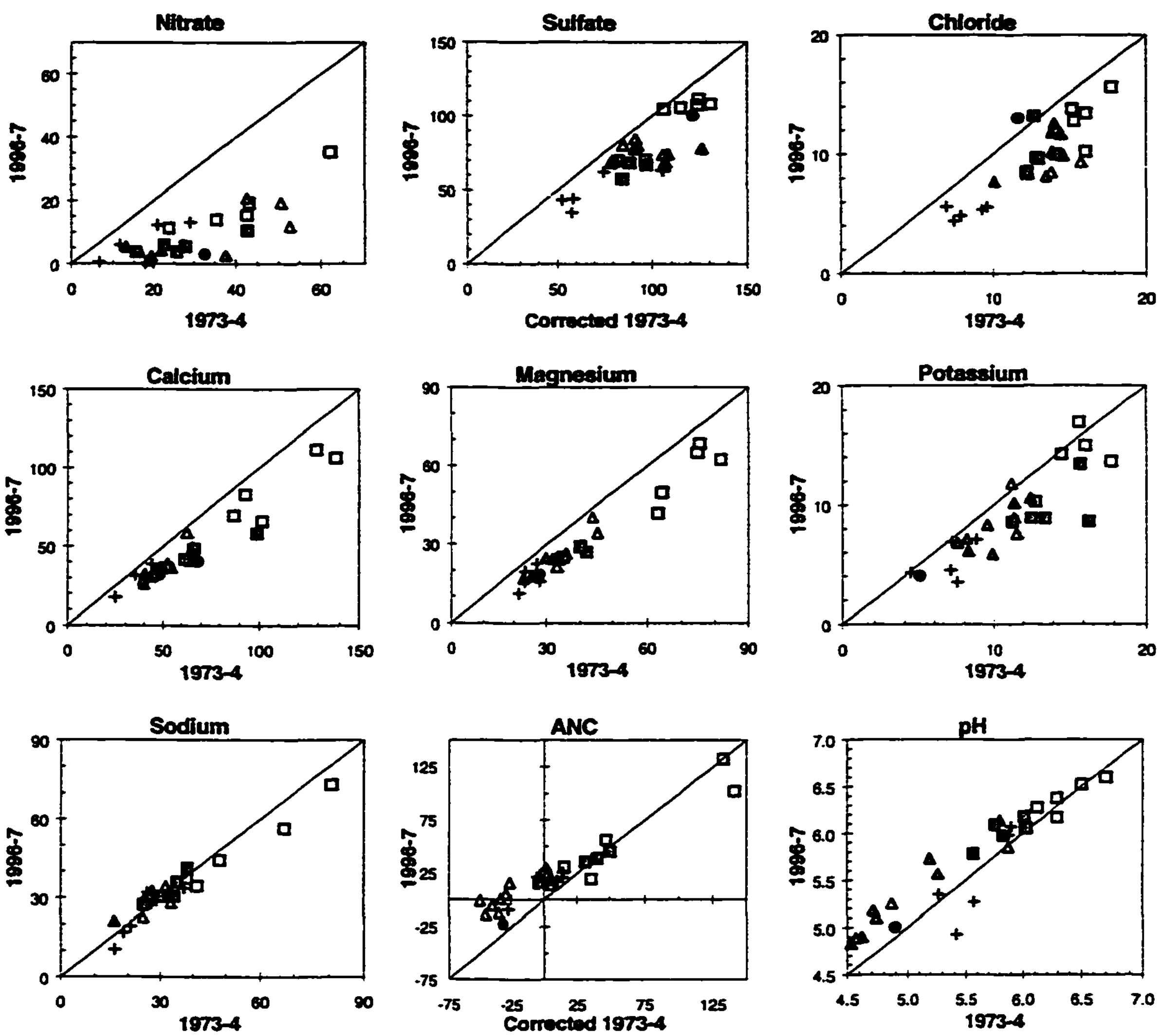

Figure 3.5: Stream-by-stream comparisons of annual 1996-7 ion concentration (μeq/L) or pH with 1973-4 values. Distances above or below the 1:1 lines indicate increases or decreases from 1973-4 values. The watersheds on Mt. Washington (+) generally face west and have spruce-fir vegetation, while spruce-fir (Δ) and hardwood (□) forests occur on both the west (open symbols) and east (closed symbols) side of Mt. Moosilauke. Changes at Hubbard Brook Watershed 6 between 1973-4 and 1993-4 are included for comparison (•). Note the different scales for different ions.

Nitrogen. Every stream had lower NO_3^- concentrations at every sampling period in 1996-7 relative to 1973-4. Nitrate concentrations decreased across all Mt. Moosilauke streams, regardless of forest type or successional stage. However, spatial trends among successional stages persisted in 1996-7. During both sampling years, NO_3^- concentrations of streams draining old-aged stands with no known logging history exceeded those of successional stands logged in 1943-1947 (Figure 3.6).

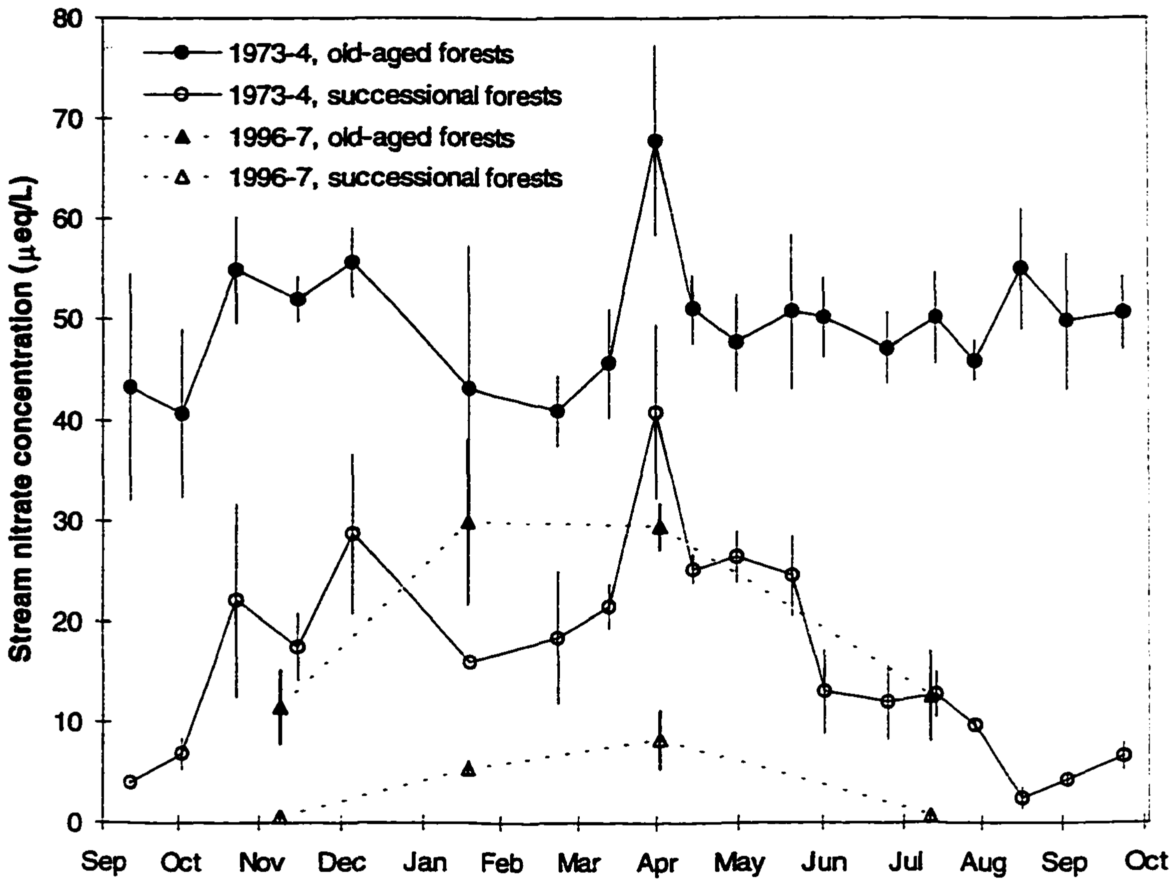

Figure 3.6: Stream nitrate concentrations in 1973-4 (—) exceeded those in 1996-7 (---) for both old-aged (solid symbols) and successional (open symbols) forests. Each point represents the mean (± SE) of four streams draining successional forests or seven streams draining old-aged forests.

Base Cations. In both years, concentrations of all base cations were higher on the western side of Mt. Moosilauke than on the eastern side, possibly resulting from differences in till chemistry across the mountain. However, the declines in Ca^{2+} and Mg^{2+} concentration between the two years did not vary by aspect, but declined more in streams draining hardwood stands than in those draining spruce-fir. Cation concentrations were higher in hardwood streams than spruce-fir streams, so that proportional declines were consistent between vegetation types.

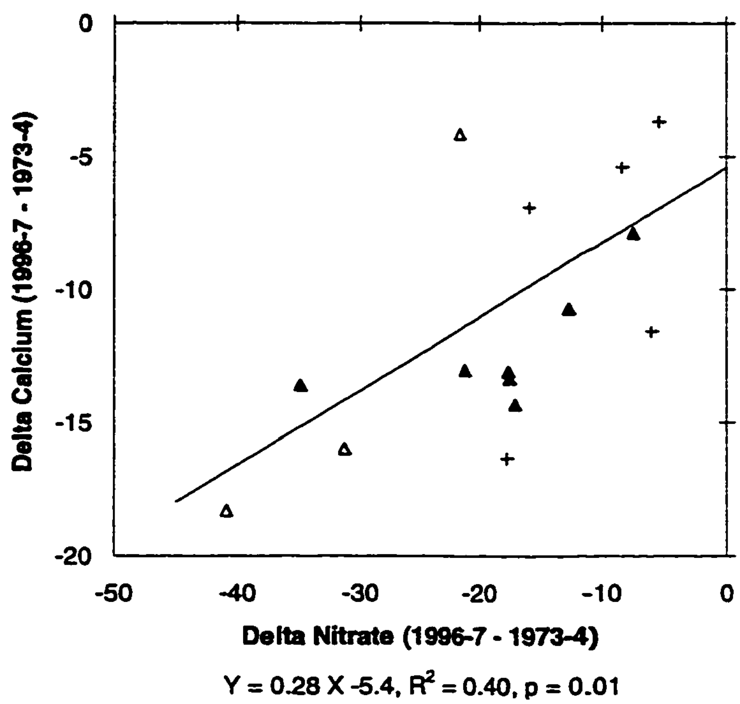

Figure 3.7: In spruce-fir watersheds (n=15), decreases in Ca^{2+} concentration (μeq/L) correlated with decreases in stream NO_3^- (μeq/L). Symbols as in Figure 3.5.

There was only one significant correlation between changes in individual or total base cations and changes in individual or total anions. Multiple regression analyses indicated that the decreases in Ca^{2+} concentration correlated with decreases in NO_3^- concentration ($P = 0.05$) but not with decreases in corrected SO_4^{2-} concentration ($P = 0.38$). The calcium-nitrate relationship was statistically significant but poor (n = 25, $P = 0.05$, $R^2 = 0.15$) when considered across all streams, but stronger when examined in spruce-fir catchments, only (Figure 3.7; n = 15, $P = 0.01$, $R^2 = 0.40$).

ANC and pH. Between 1973-4 and 1996-7, pH and ANC in high-pH streams changed little, while pH and ANC of streams that were acidic in 1973-4 generally increased (Figure 3.5). Mean ANC values were -25 μeq/L in 1973-4 in Mt. Moosilauke spruce-fir stands, suggesting that hydrogen and aluminum were leached to maintain charge balance. In 1996-7, mean ANC increased to 5 μeq/L, indicating a partial recovery of ANC in the spruce-fir stands.

While the calculated changes in ANC may be suspect due to potential problems with the 1973-4 SO_4^{2-} data, the changes in adjusted ANC were highly correlated with changes in pH in the Mt. Moosilauke streams (Figure 3.8). Two Mt. Washington streams decreased in mean pH despite increasing ANC, which suggests possible errors in measurement or stream relocation.

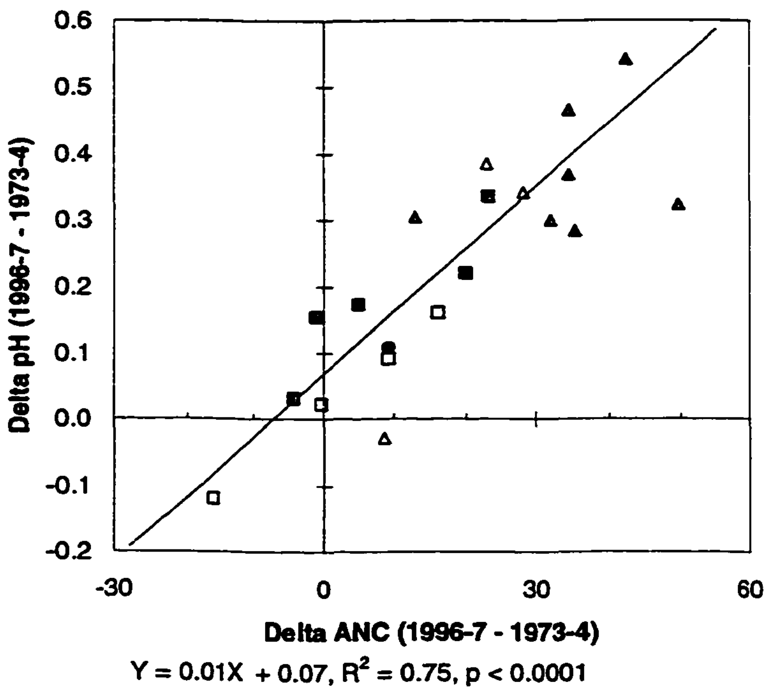

Figure 3.8: Among all Mt. Moosilauke streams (n=20), increases in stream pH correlated with increases in ANC (μeq/L). Symbols as in Figure 3.5.

On Mt. Moosilauke, multiple regression analysis indicated that changes in mean annual pH can be partially explained (n = 20, $P < 0.001$, Adj. $R^2 = 0.72$) by changes in base cation ($P < 0.001$), SO_4^{2-} ($P = 0.001$), and NO_3^- concentration ($P = 0.04$). The regression coefficients indicated that pH declined by 0.07 pH units for every 10 μeq/L decline in base cation concentration, and increased 0.07 and 0.08 pH units for every 10 μeq/L decline in NO_3^- and corrected SO_4^{2-} concentration, respectively.

Discussion

Before attributing changes in stream chemistry to long-term changes in deposition or forest nutrient cycling, other potential causes must first be addressed. Analytical methods, streamflow, or sampling frequency may have caused differences between 1973-4 and 1996-7 stream chemistry for reasons unrelated to changes in deposition or forest succession.

Alternative Causes for Changes in Stream Chemistry

Analytical Methods. Some of the analytical methods used in 1996-7 have greater precision and lower detection limits than those used in 1973-4 (Table 3.1). Of all ions, SO_4^{2-} displays the least consistent changes over time, possibly due to the poor precision of the turbidimetric analysis used in 1973-4. However, the correlation of declining pH with declining SO_4^{2-} concentrations suggest that the sulfate measurements may be imprecise, but qualitatively representative.

In 1973-4, NO_3^- concentrations were measured by reduction to NH_4^+, conversion to NH_3 with NaOH, and use of an ammonia electrode. Nitrate standards were included to confirm complete conversion of NO_3^- to NH_3, but the ammonia electrode can overestimate low NO_3^- concentrations by 3-6 μeq/L (Greenberg et al. 1992). This bias can not explain the large decreases in NO_3^- concentration between 1973-4 and 1996-7.

Streamflow. Did 1996-7 samples have lower ion concentrations because they were diluted by greater streamflow than during 1973-4? Daily streamflow records at HB W6 indicate that flows on sampling dates in 1996-7 were similar to those in 1973-4. The Mt. Moosilauke watersheds are generally small, steep, catchments subject to the same general weather patterns as HB W6. Median daily (1.4 mm) and total annual (1159 mm) streamflow in 1973-4 exceeded median (1.0 mm) and total (1046 mm) flows in 1996-7, and both years exceeded long-term (1963-1988) annual average flows (896 mm; Federer et al. 1990). HB W6 streamflow was slightly greater on sampling dates in Jan. 1974 (1.8 - 2.4 mm) than on those in Jan. 1997 (0.7 and 0.8 mm), while streamflow in July 1997 (0.8 mm) exceeded July 1974 (0.1 - 0.4 mm). Nov. and April flows were variable in both years, but ranges overlapped. The streamflow differences between years are small and variable in sign, and do not introduce enough bias to have caused the consistent declines in SO_4^{2-}, NO_3^-, Ca^{2+}, and Mg^{2+} concentration.

Seasonal Biases. The observed differences in Mt. Moosilauke stream chemistry could be artifacts of the sampling scheme used to characterize 1973-4 and 1996-7 annual averages. Despite the strong correlation between quarterly and monthly averaged ion concentrations in other White Mountain streams (Figure 3.3), the quarterly sampling schedule may not have adequately characterized annual 1996-7 stream chemistry on Mt. Moosilauke. In 1973-4, a greater proportion of the annual samples were collected during the growing season than during 1996-7. Since NO_3^- concentrations are generally lowest during the growing season and highest during the dormant season, the bias introduced by the sampling schedule would underestimate 1973-4's true mean NO_3^- concentration and overestimate 1996-7's. Despite this possible bias, NO_3^- concentrations decreased substantially at all streams.

Biogeochemical Causes

Comparison with Long-term Records. The observed changes in stream chemistry on Mt. Moosilauke and Mt. Washington across two decades are supported by the long-term records at Hubbard Brook which shows similar changes over a similar time period (Figure 3.5). The HB W6 record indicates that SO_4^{2-}, Ca^{2+}, and Mg^{2+} concentrations have declined gradually and relatively consistently over the past three decades, while NO_3^- concentrations have declined erratically since their peak in 1969-1976 (Driscoll et al. submitted; Likens and Bormann 1995). The early 1990s have the lowest NO_3^- concentrations in the HB W6 record (Driscoll et al., submitted).

Declining Sulfate Deposition. Over the past 1-2 decades, SO_4^{2-} concentrations of surface waters have declined across the northeastern U.S. (Stoddard et al. 1998a, b), in Adirondack, New York lakes (Driscoll et al. 1995), Catskill, New York streams (Murdoch and Stoddard 1993), Vermont lakes (Stoddard and Kellogg 1993), Massachusetts streams (Mattson et al. 1997) and Maine lakes (Kahl et al. 1993). The declines in surface water SO_4^{2-} concentration have been attributed to declining sulfur emissions resulting from the Clean Air Act of 1970 and its amendments in 1990.

Declining Base Cation Concentrations. Declines in surface water base cation concentrations have been reported at most northeastern U.S. monitoring sites (Driscoll et al. 1989, 1995, Likens et al. 1995, 1998, Stoddard and Kellogg 1993, Stoddard et al. 1998a). Two main mechanisms have been proposed to account for the observed declines

(Driscoll et al. 1989). First, reduced acid loading requires that fewer positively-charged ions be leached to maintain charge balance (Galloway et al. 1983, Reuss & Johnson 1985); second, deposition of base cations has declined over the eastern United States over the past two decades (Hedin et al. 1994). Either or both of these mechanisms may have caused the declines in Ca^{2+} and Mg^{2+} concentration observed in the resampled streams.

Changing pH and ANC. Stoddard (1998a) reported that across New England, the greatest increases in ANC have occurred in lakes that previously had the lowest ANCs. Similarly, on Mt. Moosilauke the most acidic streams had the largest increases in ANC and pH. The high elevation spruce-fir catchments generally have thin, poorly buffered soils, making them particularly sensitive to changes in acid inputs.

The long-term recovery of stream pH and ANC will depend on continued reductions in SO_4^{2-} deposition, particularly in response to reductions in base cation deposition. While surface water SO_4^{2-} concentrations may continue declining over the next several decades, NO_3^- concentrations cannot. The NO_3^- concentrations 1996-7 were close to zero in many streams and hence are likely to increase at least sporadically in response to climate variation, continued N deposition, and aging forests (Aber and Driscoll 1997). In Catskill, NY streams, NO_3^- concentrations have increased as SO_4^{2-} concentrations have decreased, and so few changes in pH or ANC have occurred (Stoddard 1991, Murdoch and Stoddard 1992; 1993).

Declines in Stream Nitrate. Why has stream NO_3^- declined in the White Mountains? Several theories have been advanced to explain both the NO_3^- peak in the 1970s and the drop in the 1990s, including changes in N deposition, insect outbreaks, soil frost, and interannual climate fluctuations.

Changes in wet deposition do not readily explain the sharp declines in stream NO_3^- concentration. At Hubbard Brook, there are no significant long-term trends (1965-1992) in either the concentration or flux of inorganic N in wet deposition (Driscoll et al. submitted, Likens and Bormann 1995). In the Adirondacks, small declines in wet deposition NH_4^+ (-0.57 μeq L^{-1} yr^{-1}) and NO_3^- (-0.13 μeq L^{-1} yr^{-1}) concentration have been detected over 1978 - 1993 (Driscoll et al. 1995), while no changes have been observed in the Catskills (Stoddard 1991) or across the northeast as a whole (Stoddard et al. 1998b).

Insect infestation and defoliation can raise stream NO_3^- concentrations by reducing plant demand and transferring plant N to the soil (Swank et al. 1981, Eshleman et al. 1998). Eshleman et al. (1998) recently suggested that the high NO_3^- concentrations observed at Hubbard Brook during the early 1970s may have been due to heavy insect defoliation reported in 1969-70 (Bormann and Likens 1979). However, the Mt. Moosilauke and Mt. Washington data indicate that 1973-4 NO_3^- concentrations were relatively high in all streams across the White Mountain region, in both spruce-fir and northern hardwood-dominated watersheds. It is unlikely that insect outbreaks occurred simultaneously in both vegetation types on both mountains.

Soil frost may trigger losses of NO_3^- to streams by disrupting the soil structure and by lysing nitrogen previously held by soil microbes. Mitchell et al. (1996) observed synchronous patterns of peak NO_3^- losses in streams across the northeastern U.S. following unusually cold temperatures in the winter of 1989-90. Likens & Bormann (1995) noted that high NO_3^- losses at Hubbard Brook in early 1970 and 1974 coincided with widespread soil frost, and Fahey and Lang (1975) observed concrete frost across Mt. Moosilauke during the late fall and winter of 1973-4. While soil frost may partially explain the high NO_3^- concentrations observed in late 1973 and 1974, it does not explain the elevated NO_3^- concentrations in the summer and fall of 1973 (Figure 3.6). Soil frost may be one of several climate-related factors contributing to interannual NO_3^- fluctuations, but additional factors must also be involved.

Using the PnET-CN model, Aber and Driscoll (1997) demonstrated that variation in monthly temperature and precipitation can explain much of the interannual variability in NO_3^- leaching at Hubbard Brook. Both plant uptake and N mineralization respond to temperature and moisture conditions, so that subtle differences in the rate or timing of biotic responses may lead to either NO_3^- leaching or N retention. Model runs suggested that severe drought at Hubbard Brook in the 1960s may have led to elevated NO_3^- leaching in the 1970s and depleted N stores in the 1980s. However, the model failed to predict the extraordinarily low NO_3^- losses at HB W6 in the early 1990s. Because N is so biologically active, streamwater N losses may be highly sensitive to small changes in biological processes that are not explicitly modeled. Increasing atmospheric CO_2 concentrations could be increasing plant uptake of N, but no regional increases in growth have been reported. The nearly uniform declines in NO_3^- concentrations across all resampled streams suggest a regional controller such as atmospheric chemistry or climate.

Forest Succession and Nitrogen Saturation. The decline in NO_3^- concentration over 23 years appears to contradict current conceptions of both N saturation and successional changes in N retention (Driscoll et al. submitted). Despite chronic N deposition and two decades worth of forest maturation, NO_3^- leaching has decreased at all sites. Not only have NO_3^- concentrations declined, but seasonal patterns appear to have shifted as well (Figure 3.6). In 1973-4, NO_3^- concentrations in streams draining old-aged stands displayed little seasonal variation, yet in 1996-7 the same streams appear to have dormant-season maxima and growing-season minima. These changes in seasonal patterns correspond with a shift in N saturation status from the high, aseasonal NO_3^- leaching of late stage 2 to a late stage 1 pattern of seasonal trends and slightly elevated base flow (Stoddard 1994).

Interannual climate variation may mask both progressive N saturation and successional declines in N retention. Climate-induced variability in annual NO_3^- leaching can prevent the detection of chronic N deposition for several decades (Aber and Driscoll 1997). Similarly, the large year-to-year difference in forest N uptake due to climate variation likely exceed any small declines in forest uptake due to two decades worth of aging. However, within a particular year, successional status can account for substantial differences in NO_3^- leaching among watersheds (Figure 3.6; Leak and Martin 1975, Vitousek and Reiners 1975, Pardo et al. 1995, Aber and Driscoll 1997, Chapter 1). The old-aged stands on Mt. Moosilauke continue to have higher NO_3^- losses than recovering successional stands. Similarly, NO_3^- leaching from the Bowl, a nearby old-growth hardwood watershed, exceeded that from the successional forests of Hubbard Brook both in the early 1970s (Martin 1979) and in the early 1990s, although NO_3^- fluxes were lower at both sites in the 1990s (Aber and Driscoll 1997, Driscoll et al. submitted, Martin et al. in prep.).

It appears that some combination of climate factors has led to the current decline in NO_3^- leaching across much of New England, overriding any signal of chronic N deposition on forests that have matured over the past few decades. However, even current models of climate and NO_3^- leaching cannot fully account for the low NO_3^- losses in the 1990s. Since 1974, at least 10.7 keq/ha N have fallen on the southwestern White Mountains in wet deposition alone. If interannual climate variation does indeed explain the observed declines in NO_3^- leaching since 1973-4, then future climate variation or climate change could cause NO_3^- losses to match or exceed those observed in the 1970s.

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APPENDIX 1

STREAM LOCATIONS

Appendix 1.1. Land-use History Streams: Forest History and Location

Site 1: George's Gorge, Pinkham Notch

History. The upper portions of these watersheds were owned by the Conway Lumber Company, while the Umbagog Realty company owned the lower slopes adjacent to the present-day Rt. 16. A 1911 map of the Conway Lumber Co. tract by F. A. Gardner included a small burned patch on the ridge due north of the AMC Pinkham Notch camp. In a 1914 report on the lands of the Conway Lumber Co., D. W. Martin did not mention the small burn, but he did include it in his 1915 map and 1916 report on the Umbagog tract. The 280 acre burned area, currently crossed by the Crew Cut and George's Gorge hiking trails, was reported to support regrowth of small cherry and birch. Tree cores from four different trees on the site indicate that they reached breast height in 1907 (paper birch), 1908 (bigtooth aspen), 1910 (yellow birch), and 1936 (paper birch), respectively. Allowing 3-4 years to reach breast height, the fire likely occurred around 1903, a year when many fires occurred across the region (Chittenden 1904).

Stream 1A. George's Gorge, sampled from the George's Gorge hiking trail, ~10 m above its junction with the Crew Cut trail.

Stream 1B. The next stream crossed by the Crew Cut trail beyond George's Gorge, northeast of stream 1A. Occasionally dry.

Stream 1C. Peabody River tributary. The lower portion of this watershed was likely logged and may have burned, but the upper portion, draining subalpine fir and possibly the alpine region near Nelson Crag, likely did not. This stream was included as mixed land-use history subalpine/logged. This is the first stream crossed on the Crew Cut Trail after leaving the Old Jackson Road, not to be confused with a stream that flows through this junction.

Site 2: Little Wildcat

History. Chittenden (1904, p. 38) reported that the forests of the Carter Range had been heavily lumbered by the time of his survey. The 8540 acres on the western slopes of the Carter Range were owned by the American Realty Company, a subsidiary of International Paper. In a 1916 survey, D. W. Martin reported simply that "practically all of this tract has been logged over for the softwoods during the last twenty years so that at present, there is only about 645 acres of virgin timber left." Later, he stated that "these lands originally supported an excellent stand of spruce and fir but have been logged almost continuously for the last 30 years, so that at present time, all merchantable timber left is to be found high up on the mountain and in very rough and inaccessible places." Specifically focusing on the region near the current Wildcat Ski Area, Martin reported that the stands had been "stripped. No merchantable timber of any kind remains. The small area between Thompson Brook and the southern part of the tract has also been cleared of all merchantable timber, except a fringe along the edge of the ridge."

Stream 2A. Little Wildcat Brook, where it is crossed by Rt. 16, just south of the first turnoff north of Wildcat Ski Area. Sampled ~10 m upstream to avoid road salt influence.

Stream 2B. Small stream approximately 150 m north of Little Wildcat, on the east side of Rt. 16 between the first and second pullouts north of Wildcat Ski Area. Sampled ~5 m upstream to avoid road salt influence.

Site 3: Lost Pond, Pinkham Notch

History. Chittenden (1904) reported that "the slopes draining into the Glen Ellis River, from the village of Jackson to Wildcat Mountain, have been stripped of all coniferous growth (p. 38)." Most of these watersheds fell in the Umbagog Realty holdings in Pinkham's Grant, while the upper portions may have been owned by American Realty. Both holdings were heavily logged, marked as "cutover" in the 1915 map by D. W. Martin, who reported in 1916 that these cutover lands contained no merchantable timber.

Stream 3A. Small tributary to the Ellis River. Crossed by the first small, 2-3 log bridge on the Lost Pond Trail after leaving Rt. 16. Opposite a large pool in the Ellis River.

Stream 3B. Small tributary to the Ellis River. Crossed by the second small 2-3 log bridge on the Lost Pond Trail after leaving Rt. 16, approximately 150 m south of stream 3A.

Site 4: Glen Boulder

History. This stand comprised the southern-most portion of a tract owned by the Umbagog Realty company, on the far side of Pinkham Notch from the rest of the tract. In his 1916 report, D. W. Martin indicated that "in the past this tract, with the exception of 435 acres of virgin timber, has been logged for all softwoods." He later tallied 165 acres of virgin hardwoods and spruce, and 270 acres of virgin spruce. His 1916 map of

Pinkham's Grant indicated that most of the tract south of Glen Boulder and the Glen Ellis falls was either virgin spruce of virgin hardwoods and spruce:

> *The virgin timberland is divided into virgin spruce and virgin hardwoods and spruce. The greater part of it, however, is on very steep ground adjacent to Glen Ellis Falls. The hardwood and spruce supports a scattering of good spruce among some large old growth hardwoods... logging would be expensive. The ground is steep and far from market.*

Stand compartment records indicate that small patches of salvage cut occurred immediately adjacent to Rt. 16 in 1959 after a blowdown. On-site inspection suggests that an old path or road ran through the lowest portion of the tract, parallel to the present Rt. 16. Road-building materials may have been excavated from a large pit south of stream 4A and north of stream 4B.

Stream 4A. Slide Brook, crossed by Rt. 16 with a large bridge labeled "E-1." This watershed does not appear to support old-growth vegetation, and the land's geomorphology suggests that it may have channeled the course of past landslides (hence the name). A more recent slide may have occurred on the upper watershed in 1969 (D. Bryant, personal communication). This stream was not included in analyses as old-growth, but as possible landslide / mixed land-use history.

Stream 4B. The left (southern-most) of two streams joining at the forest edge at the top of the Rt. 16 embankment. Sampled ~ 10 m in from the forest edge.

Stream 4C. Similar to stream 4B. Sampled ~ 10 m in from the forest edge.

Stream 4D. The first stream north of the road marker distinguishing Pinkham's Grant from Jackson Township. This watershed has more spruce and hemlock than streams 4B and 4C. Sampled ~20 m from the forest edge at the top of the Rt. 16 embankment.

Site 5: Rocky Branch

History. The upper portion of these watersheds were owned by the Conway Lumber Company, while the lower slopes were owned by E. Libby and Sons. The 1911 map of the Conway Lumber Company lands by F. A. Gardner does not indicate any burned areas in the region of the watersheds. D. W. Martin, in his 1914 report on the lands owned by the Conway Lumber Company indicated that heavy cutting had occurred in the Rocky Branch River drainage over the previous eight years and that:

> *...the ground is at present covered with the recent slash, and is easily inflammable, so that the fire risk in this valley is very great. The burned territory in the Rocky Branch is the result of two fires, one in 1912, and the other 3,200 acres in 1913 and 1914.*

The tract owned by E. Libby and Sons was mapped by E. A. Morrison and party in 1934. They mapped the northern portion of the tract as "unmerchantable burn" with vegetation in a 0-20 year age class.

Stream 5A. First streambed crossed by the Rocky Branch Trail, approximately 3/4 mile from the trailhead on Rt. 16. Dry in late summer / early fall.

Stream 5B. Second stream encountered on the Rocky Branch Trail, approximately 100-200 m north of stream 5A.

Stream 5C. A large stream crossed by the Avalanche Brook Ski Trail, which joins the Rocky Branch Trail ~20 m below stream 5C.

Stream 5D. A small stream crossed by the Avalanche Brook Ski Trail, less than 20 m north of stream 5C.

Site 6: Gibbs Brook, Crawford Notch

History. The Gibbs Brook watershed supplied drinking water to the former Crawford House, one of the White Mountain Grand Hotels. The site was never clearcut, and is currently a Scenic Area and candidate Research Natural Area. The 1901-2 Report of the New Hampshire Forestry Commission indicates that "this area is further protected by the wise action of the hotel proprietors. The owners of the Crawford House hold large tracts, bought to control forest cutting (p. 24)".

These forests may have a high internal disturbance regime due to northwest exposure and reported red spruce decline (Foster and Reiners 1983, Lawrence et al. 1997). Foster and Reiners (1983) described the structure of spruce-fir forests but noted extensive spruce mortality. Personal observation indicated that large dead spruce boles were common, and hobblebush was noticeably thick. Foster and Reiners (1983) did not report evidence of old fires at the site, yet I observed frequent occurrences of charcoal on the lower portion of the watershed of Elephant Head Brook, along the Webster-Jackson Trail. Charcoal occurred below the forest floor or beneath exposed roots of large, old spruce and hemlock. No charcoal was found on the three other watersheds.

Stream 6A. Gibbs Brook. Sampled at the bridge marking the junction of the Crawford Path and a spur trail to a WMNF parking area on the Mt. Clinton Road.

Stream 6B. A tributary to Gibbs Brook. The stream is diverted below the Crawford Path in a metal culvert, the first such structure encountered along the trail upstream from the bridge. Sampled ~ 5 m above the culvert.

Stream 6C. Elephant Head Brook. Sampled at a small bridge ~15 m upstream of the brook's junction with the southern end of Saco Lake.

Stream 6D. A tiny tributary to Saco Lake, entering on the lake's east edge, just north of a lookout rock. Drains a small cove of large yellow birch and sugar maple. Sampled ~ 30 m upstream of the lake.

Site 7: Mt. Washington

History. Forests on these watersheds were owned by the Conway Lumber Company. The 1911 survey map by F. A. Gardner indicated that the forests covering the small watersheds of streams 7B and 7C were cutover (heavily logged). The survey indicates that the lower portion of the Clay Brook watershed was also cutover prior to 1911. Both stand compartment records and tree rings indicate that the remaining forests were impacted by the 1938 hurricane. The survey map indicated that the subalpine forests above the sampling points on the Monroe (7D) and Ammonoosuc (7E) Brooks were not logged. Compartment records do not indicate cutting in any of these watersheds since acquisition, although some of the fir stands below the sampling point on Clay Brook were authorized for harvest in 1967.

Stream 7A. Clay Brook. Sampled along the Jewell Trail, ~ 5 m downstream of the bridge crossing the brook. Included as mixed alpine/subalpine/logged land-use history.

Stream 7B. The only significant stream encountered along the Jewell Trail before reaching Clay Brook. Classified as logged land-use history.

Stream 7C. A tiny stream crossing the Ammonoosuc Trail shortly below a junction with a spur trail that connects to a WMNF parking area. Classified as logged land-use history.

Stream 7D. Monroe Brook. The largest stream crossed by the Ammonoosuc Trail before reaching Gem Pool. Classified as subalpine/alpine land-use history.

Stream 7E. Ammonoosuc Brook, sampled at Gem Pool, the large pool crossed by the Ammonoosuc hiking trail before it begins to ascend steeply to Lakes of the Clouds. Classified as alpine/subalpine land-use history.

Site 8: Zealand Valley

History. Chittenden (1904) and the 1901-2 Report of the New Hampshire Forestry Commission indicate that this fire occurred in 1888, a claim that Belcher (1980) convincingly disputes and dates to 1886 based on several period accounts. In particular, Belcher cites a report by Frank H. Burt in *Among the Clouds*, recounting an 1887 trip through the Zealand Valley:

> *The lower part of the valley is rather dreary. The spruce has been cut, and the great fire of 1886 destroyed all that the woodsman had left. On the hillsides in many places, even the soil had yielded to the intense heat and nothing was left but the bare rock.*

The 15,045 acre tract had been owned by J.E. Henry & Sons, and was surveyed in 1915 by D.W. Martin. Martin reported that:

> *The greater portion of this tract, about seventy percent, was burned subsequent to the logging operations about twenty years ago. This area has sprung up to a growth of popple and birch, and the more favorable places it has attained merchantable size at the present time. This area has been designated as merchantable second growth... On the upper slopes the timber on large areas of this burned land has not reached merchantable size, and in many places the fire was so severe that practically all tree growth was destroyed, and there is a very straggling reproduction at present.*

Compartment records indicated that forests on all four sampled watersheds were slated to have been thinned between 1984 and 1988 (the "Hot Shot Sale"), although none of these stands were included in a 1988 summary of thinning activities. A 1996 survey indicated that most of the stands marked for thinning in 1984 were in need of thinning in 1996.

Stream 8A. All four streams were sampled along the Zealand Valley Rd. Stream 8A is 1.1 mi. from Rt. 302, in a distinct drainage encountered south of the Sugarloaf Campground and beyond the end of paved road.

Stream 8B. Streams 8B, 8C, and 8D are within 200 m of each other. Stream 8B is a very small stream that sluggishly drains a damp forested stand 1.8 mi. from Rt. 302.

Stream 8C. Sampled the right (north-) most stream of two converging before entering a culvert beneath the Zealand Rd; approximately 1.85 mi. from Rt. 302.

Stream 8D. Hale Brook. A large, broad streambed encountered at 1.9 mi. from Rt. 302.

Site 9: Mt. Bickford

History. Chittenden's 1904 map of forest condition in northern New Hampshire indicates that much of the northern slope of Mt. Lafayette and Mt. Garfield burned in the 1903 fires, but he did not discuss the condition of these particular forests. A 1913 survey by D. W. Martin and others confirmed that this area had burned. Stand compartment records do not indicate that any harvesting activities have occurred on the western half of the Skookumchuck Brook watershed since acquisition by the federal government. Thinning and patch cuts have occurred on the eastern half of the watershed over the past two decades.

Stream 9A. Streams 9A, 9B, and 9C all drain under the bike path in culverts, and were all sampled ~ 10 m upstream of the bike path. Stream 9A is the furthest from the Governor Galen circle (see site 10), the last small stream encountered before reaching

Skookumchuck Brook. Charcoal was found in nearby soils, and an old logging road follows the eastern side of the stream.

Stream 9B. This stream was diverted under the bike path in a culvert. This stream was encountered shortly before reaching stream 9A.

Stream 9C. This stream drains into the first culvert / tunnel encountered along the bike path on the eastern side of the large bridge over Lafayette Brook. Not to be confused with a small wet area draining into what appears to be a cement block at the start of the bike path.

Stream 9D. A small, temporary stream that enters Lafayette Brook shortly before the brook is crossed by the large bridge and the bike path. The stream is encountered immediately after entering the woods on an old access path on the east side of Lafayette Brook.

Site 10: Lafayette Brook

History. The 116 acre tract on the north slope of Mt. Lafayette was formerly owned by Publisher's Paper Co. Surveys in 1915-1917 indicated that:

> *The tract consists principally of subalpine and barren with a small patch of virgin spruce on the north side. This timber is rather scrubby and intermixed with short paper birch. No part of this tract has ever been cut. The topography is very steep and rough.*

A summary by the New Hampshire Natural Heritage Inventory (Sperduto and Engstrom 1993) indicated that the site contained old-aged northern hardwoods on the lower slopes, grading to spruce and yellow birch with elevation. They noted frequent snags in various stages of decay, and trees in a range of size classes. Some spruce reached 75 cm dbh. Compartment records indicated that no cutting had occurred on the watershed since federal acquisition.

Stream 10A. The first deep culvert encountered after turning into the access road for the Governor Galen Memorial, exit 3 off Rt. 93 through Franconia Notch State Park. Descend the embankment just before the "Highway Ends 1000 ft." sign. Sampled ~15 m upstream of access road.

Stream 10B. The second drainage encountered on the access road. Sampled ~ 5 m upstream of road. Stream dried in late summer.

Stream 10C. Drains in a large culvert encountered just before reaching the parking circle. Sampled ~30 m. upstream of road in an effort to find flow. Stream flowed only during spring, and was not included in any analyses.

Site 12: Nancy Brook

History. The stands now comprising the Nancy Brook Research Natural Area were once owned by Daniel Saunders & Company, a logging company commended by both Chittenden (1904) and Belcher (1980) for its policy of selective cutting. In a 1933-4 survey of the Livermore tract, A. Morrison and company noted extensive areas of virgin spruce in the upper Nancy Brook and Halfway Brook drainages. However, they classified the stands directly surrounding Nancy Pond as 40-80 or 80-120 year-old spruce. Compartment records indicated that the 1938 hurricane extensively damaged portions of the old-growth stands, and the area below Nancy Cascade was salvage logged between 1939 and 1943.

Stream 12A. Nancy Brook. Sampled at the base of Nancy Cascade, 2.6 miles from Rt. 302 on the Nancy Brook Trail. This brook drains Nancy Pond as well as a small pond high on the col between Mt. Nancy and Mt. Bemis.

Stream 12B. A tributary to Nancy Brook, entering shortly below (south of) Nancy Cascades. Sampled upstream of this junction, at the upper portion of the bowl formed by streams 12A & B and the steep slope to the west. This stream drains the slopes of Duck Pond Mountain, and appears free from influence by Nancy Pond or any other ponds.

Table A1.1 GPS locations of stream sampling points - land-use history streams.

Stream	Elev. MSL (m)	Latitude	Longitude	NH State Plane (m) 1983 datum Northing	Easting	SD (m)
01A	637.4	44.26422	-71.25137	196082.9	333160.1	0.91
01B	643.7	44.26494	-71.25014	196163.1	333258.2	1.01
01C	632.1	44.26301	-71.25312	195947.7	333020.7	0.43
02A	533.3	44.27412	-71.23171	197190.3	334724.2	0.68
02B	533.0	44.27534	-71.23133	197326.8	334753.9	1.24
03A	615.4	44.25383	-71.25172	194928.1	333138.0	0.94
03B	614.2	44.25232	-71.25137	194760.1	333166.6	1.15
04A	557.7	44.23850	-71.25667	193223.1	332751.3	0.46
04B	542.7	44.23591	-71.25766	192934.3	332673.4	1.17
04C	539.6	44.23497	-71.25766	192830.0	332674.2	3.50
04D	534.1	44.23389	-71.25782	192709.6	332662.0	0.58
05A	508.5	44.20623	-71.24808	189640.3	333455.2	0.79
05B	519.0	44.20655	-71.24901	189675.0	333381.1	1.75
05C	532.8	44.20700	-71.25055	189725.2	333257.9	1.00
05D	540.1	44.20725	-71.25092	189752.7	333228.3	0.52
06A	640.4	44.22101	-71.40653	191230.0	320786.3	0.78
06B	669.8	44.22077	-71.40429	191203.8	320965.3	0.42
06C	583.5	44.21634	-71.40853	190710.7	320628.1	1.35
06D	586.4	44.21838	-71.40825	190937.2	320650.1	0.94
07A	875.6	44.27490	-71.34669	197235.5	325544.3	1.08
07B	887.6	44.27337	-71.34715	197065.5	325508.0	1.55
07C	872.9	44.26924	-71.34506	196607.0	325676.9	0.37
07D	982.6	44.26671	-71.33327	196329.3	326619.3	0.55
07E	1048.9	44.26755	-71.32687	196424.3	327130.5	0.28
08A	511.1	44.24923	-71.50226	194346.7	313130.7	0.87
08B	522.3	44.24255	-71.49357	193605.4	313826.4	1.17
08C	526.0	44.24236	-71.49295	193584.2	313875.9	0.96
08D	530.7	44.24197	-71.49179	193541.0	313968.7	0.74
09A	557.9	44.18905	-71.67926	187645.8	298992.8	0.89
09B	572.4	44.18797	-71.68014	187526.6	298923.0	3.87
09C	570.2	44.18605	-71.68173	187313.2	298795.3	0.92
09D	575.9	44.18374	-71.68394	187056.0	298618.8	0.89
10A	586.8	44.18123	-71.68915	186778.0	298202.6	1.58
10B	588.0	44.18211	-71.68757	186875.3	298328.6	0.71
10C	582.6	44.18257	-71.68549	186926.3	298495.3	1.53
12A	733.3	44.11524	-71.38605	179483.0	322462.5	0.77
12B	723.8	44.11530	-71.38482	179490.4	322560.9	0.51

Appendix 1.2. Stream Locations - Resampled from Vitousek (1975)

Italicized text is the stream description directly from Vitousek (1975), Appendix A, which included elevations from an altimeter. Standard text provides additional or updated information from 1996-8. * indicates streams sampled in 1996-7. GPS locations and elevations (9/98) follow.

* Stream 1. *1024 m. The highest elevation stream crossing the Glencliff Trail. A small temporary stream in the spruce-fir zone with no indication of logging in the watershed.* Stream has log waterbars on either side of it, perpendicular to the trail, and four logs as mudbridges parallel with the trail directly below. Vegetation near streams 1, 2 & 3 is dominated by large dead spruce boles & skinny spruce-fir regeneration. Included in Vitousek and Reiners (1975) and Figure 3.6.

* Stream 2. *1024 m. Another small, temporary stream near #1, to which it is similar in all respects.* Stream occurs at the start of a flat section of trail, just above #3. Streambed is rocky, and is the largest above stream #5. Included in Vitousek and Reiners (1975) and Figure 3.6.

* Stream 3. *1018 m. A larger stream crossing Glencliff Trail where the trail goes through a well-developed spruce-fir stand. No logging.* One of several small, temporary, seeps in a flat section of trail, just above #4. Included in Vitousek and Reiners (1975) and Figure 3.6.

* Stream 4. *1006 m. A small temporary stream in the spruce-fir zone. No logging.* A two-log bridge crosses the streambed, next to a large rock with inch-long crystals. Gaps in the paper birch - fir forest allow slight views of the valley. This stream

is above a glade of ~50-80 yr. old fir. Included in Vitousek and Reiners (1975) and Figure 3.6.

* <u>Stream 5.</u> *930 m. One of several permanent streams crossing the trail at this point. This is the largest stream in the spruce-fir zone on Glencliff Trail. No logging.* The highest (and largest) of the many small streams in this short flat section of trail; a red spruce tree with a large bulbous deformity grows adjacent to the stream. An illegal campsite and fire ring occur just above. Included in Vitousek and Reiners (1975) and Figure 3.6.

* <u>Stream 6.</u> *899 m. A small stream appearing on the surface just above Glencliff Trail. Sampled below the trail.* The lowest of the many small streams in the short flat section of trail that ends with #5. Included in Vitousek and Reiners (1975) and Figure 3.6.

* <u>Stream 7.</u> *808 m. A small stream crossing Glencliff Trail immediately below the clearing for precipitation collection. The forest is dominated by small sugar maples in the area of collection. Some large sugar maples, indicating it was never clearcut. No evidence of any logging.* The higher of two permanent seep-streams crossing the trail just above stream #8. Many huge yellow birch and sugar maple nearby. No evidence of an old clearing for collecting precipitation. Included in Vitousek and Reiners (1975) and Figure 3.6.

* <u>Stream 8.</u> *762 m. A larger stream crossing Glencliff Trail at the only bridge on the trail. Near #7 in a similar plant community.* Still the only real bridge on the trail. Permanent stream.

* *Stream 9.* *725 m. A small stream to the southeast of Glencliff Trail, very near the trail. Hardwoods—spruce-fir transition forest. Logging history unknown, but no clearcutting.* Stream parallels, but never crosses the trail - possibly a tributary to #13.

* *Stream 10.* *716 m. A small spring-seep crossing Glencliff Trial. Large stand of hardwoods and spruce near the seep. Logging history unknown, but no clearcutting.* Seep occurs just above (northwest of) the trail, and a water bar guides it across. Can almost see #9 from here.

Stream 11. *677 m. A small permanent stream in northern hardwoods well to the north of Glencliff Trail in the northern hardwoods lysimeter site. Some large yellow birch and sugar maple, but a few stumps from logging also present. Bethlehem gneiss bedrock.* Off trail; couldn't locate.

Stream 12. *677 m. A small temporary stream near #11 and similar to it in every way.* Off trail; couldn't locate.

* *Stream 13.* *503 m. A small stream crossing Glencliff Trail immediately above a pasture. Successional birch, pin cherry, aspen and fir in the watershed. Clearcut or pastured at some time. Bethlehem gneiss and possibly Ammonoosuc volcanic bedrock.* Permanent stream occurs at the junction of the Glencliff & Hurricane Mtn. Trails, and is the water supply for the nearby Great Bear Cabin (DOC).

* *Stream 14.* *753 m. Gorge Brook where it crosses Snapper Trail. A large stream draining the east side of Mt. Moosilauke.* The Snapper Trail was rerouted in 1996 to intersect higher on the Gorge Brook Trail. The old crossing is not far below the new, and still has a log bridge which can be spotted with some difficulty while ascending the Gorge Brook Trail.

* *Stream 15. 799 m. A small stream next to Gorge Brook Trail immediately above the lowest bridge where Gorge Brook Trail Crosses Gorge Brook. Successional hardwoods vegetation.* The Snapper Trail was rerouted in 1996, and now follows #15 down to Gorge Brook. Sampled at the intersection of the new Snapper & Gorge Brook Trails, on the west side of a new bridge over Gorge Brook.

* *Stream 16. 991 m. A small stream crossing Gorge Brook Trail in a spruce-fir stand below the steep section. Within successional forest, but near its upper boundary.* The lower of two small streams crossing the trail with water bars; below "Last Water," but still on the northeast side of Gorge Brook.

* *Stream 17. 1006 m. Gorge Brook where Gorge Brook Trail begins to climb steeply away from the trail. Below the landslide terminus for landslides from South Peak.* Sampled at "Last Water," where the trail turns sharply right and away from Gorge Brook. There is a rock nearby with a memorial plaque.

Stream 18. 1305 m. A small stream high on Gorge Brook Trial. In the high elevation fir forest zone. The Gorge Brook Trail was rerouted in 1991. Multiple blowdowns and washouts meant that the old trail could not be followed with confidence.

Stream 19. 1414 m. The highest spring-seep on Beaver Brook Trail. Rarely flowing. High elevation fir forest. This portion of the Beaver Brook trail was also relocated at some point between 1974 and 1996. Not sampled.

Stream 20. 1378 m. A larger spring-seep in the high-elevation fir zone on Beaver Brook Trail.

Stream 21. 1295 m. A larger permanently flowing stream east of the boggy area in the saddle between North Peak and Blue. High elevation fir.

Stream 22. *1143 m. A small permanent stream on the steep portion of Asquamchumauke Trail. Spruce-fir and high elevation fir forest. Never logged.* The Asquamchumauke Trail was closed many years ago, but is still included on USGS maps. The lower portion of the trail is loosely maintained for DOC use, but the upper section could not be followed. Included in Vitousek and Reiners (1975).

Stream 23. *1079 m. A small stream near the base of Jobildunc Ravine on Asquamchumauke Trail. May be downstream of #22. Spruce-fir, never logged.* Included in Vitousek and Reiners (1975).

Stream 24. *1049 m. The Baker River at the fourth crossing of the Asquamchumauke Trail. Drains all of Upper Jobildunc Ravine including several beaver ponds.*

* Stream 25. *994 m. A small spring seep on Asquamchumauke Trail. Successional spruce-fir-birch, logged in 1945-1946. Stumps evident.* This small seep emerges and then disappears a few feet above the trail, immediately upslope from #26; however, this relocation may not be correct, and the true #25 could be one of several seeps above stream #26. Included in Vitousek and Reiners (1975). For streams 25-29: canopy dominated by tall skinny paper birch, with 2-5 m fir & spruce in the understory.

* Stream 26. *957 m. A small stream crossing underneath trail near #25. Same plant community.* Permanent stream with good flow. Old bridge timbers have collapsed. Included in Vitousek and Reiners (1975) and Figure 3.6.

* Stream 27. *933 m. Larger stream crossing Asquamchumauke Trail near #25 and 26. Similar plant community.* First permanent, medium-sized stream encountered when ascending the trail. Included in Vitousek and Reiners (1975) and Figure 3.6.

** Stream 28. 899 m. A small, temporary stream crossing Asquamchumauke Trail near Ridge Trial junction. Crossed by a rotting bridge. Successional fir-birch, logged in 1945-6. Stumps evident.* Second of two small, ephemeral streams crossing the trail above the intersection with the Ridge trail. A few old logs suggest an old rotting bridge. Included in Vitousek and Reiners (1975) and Figure 3.6.

* *Stream 29. 893 m. A small stream draining into the junction of Asquamchumauke Trail and Ridge Trail. Very near #28.* Temporary stream that marks the junction. Included in Vitousek and Reiners (1975) and Figure 3.6.

* *Stream 30. 869 m. Baker River just below Ridge Trail-Asquamchumauke Trail junction. Large watershed including upper Jobildunc Ravine and successional forests on the Mt. Blue ridge.* Sampled where crossed by the Ridge Trail. New bridge built in 1998.

* *Stream 31. 768 m. A small stream crossing Ridge Trail just below Camp 2. Successional spruce-fir and hardwoods.* Stream #31 has log water bars, and is not far above the junction with the Merrill Ski Trail. Nothing remains of Camp 2 but an old field.

* *Stream 32. 735 m. A large stream crossing Ridge Trail above its junction with Gorge Brook Trail. Drains most of East Moosilauke, including areas logged in 1957-8.* Believe that this is Hatch Brook, crossed by a two-log bridge with railing.

* *Stream 33. 735 m. Baker River just above its junction with #32. Near Ravine Lodge.*

* *Stream 34. 753. A small stream draining into the Ravine Lodge parking lot. Temporary stream draining the old ski area.* Enters bus turnaround. Usually dry.

* *Stream 35.* *658 m. A small stream crossing Ravine Lodge Road about a quarter mile from Rt. 118. Where four small streams join at a culvert. The second from left was sampled. Successional northern hardwoods. Kinsman monzonite bedrock.* Stand thinned ~1980 (DOC Road Sale). Also known as stream 11F.

* *Stream 36.* *634 m. Small stream crossing Ravine Lodge Road where it joins Rt. 118. Similar to #35.* Stand thinned ~1980 (DOC Road Sale). Stream #36 crosses Ravine Lodge Rd. ~10 m up from Rt. 118. Sampled above the north side of the road. Also known as stream 11E.

* *Stream 37.* *640 m. A small stream crossing under Rt. 118 and entering the DOC clearcut. Close to and similar to #35 and 36.* Stand thinned ~1980 (DOC Road Sale). The DOC clearcut has regrown. Sampled right branch of stream crossing Rt. 118 by a WMNF sign, ~ 0.2 mi. from Ravine Lodge Rd. Also known as stream 11D.

* *Stream 38.* *878 m. The first small stream crossing Jewell Trail, Mt. Washington. Spruce-fir vegetation with unknown logging history.* Stream is east of the junction with the Jewell Link (a cutoff to the USFS parking area). Also known as stream 7B.

* *Stream 39.* *872 m. Clay Brook where it crosses Jewell Trail. A large stream draining an extensive area on the west side of the Presidential Range.* Sampled below bridge crossing Clay Brook. Also known as stream 7A.

Stream 40. *1673 m. A small stream crossing the Alpine Garden Trail near its junction with the Huntington Ravine Trail. Alpine tundra, with some drainage from the Auto Road.*

Stream 41. *1661 m. Small spring-seep near the Mt. Washington cone in Alpine Gardens. Similar to #40. Above the trail.*

Stream 42. *1658 m. A spring-seep in Alpine Gardens. Also above the trail. Similar to #40.*

Stream 43. *1667 m. A large spring at the base of the Mt. Washington cone above #43. Similar to the other Alpine Garden streams.*

Stream 44. *1631 m. A small temporary stream crossing the Alpine Garden Trail near several other streams. Similar to other Alpine Garden streams.*

Stream 45. *1631 m. The largest stream in the Alpine Garden, crossing the Alpine Garden Trail close to #44.*

Stream 46. *1582 m. A small stream below Alpine Garden Trail near its junction with the Tuckerman's Ravine Trail. Considerable mountain alder near the collection site. Includes some summit drainage.*

Stream 47. *1631 m. A small stream below an alpine bog below Camel Trail. Does not drain the bog. No summit or road influences in the drainage basin. Considerable upright krummholz.*

Stream 48. *1631 m. Small stream crossing Camel trail. Drains a somewhat boggy area of tundra. No human disturbance in the watershed.*

Stream 49. *1548 m. The inlet seep to the larger Lake of the Clouds.*

Stream 50. *1554 m. A small stream draining into a cotton-grass bog off Mt. Monroe.*

Stream 51. *1295 m. A large stream crossing Ammonoosuc Ravine Trail on the headwall of Ammonoosuc Ravine. Drains much of the Clay-Monroe col but not Lakes of the Clouds. Spruce-fir, high elevation fir, and tundra in the watershed.*

* *Stream 52.* *1073 m. The Ammonoosuc River where Ammonoosuc Ravine Trail reaches the valley bottom.* Sampled at the outflow of Gem Pool, where the trail crosses the Ammonoosuc River and then ascends steeply. Vitousek may have sampled a different branch, just uphill from Gem Pool, and not crossed by the trail. Also known as stream 7E.

* *Stream 53.* *1012 m. A large stream draining Monroe and Eisenhower. An avalanche followed the stream course in the late 1960s, so while most of the watershed is in spruce-fir and fir, small pin cherries are common along the stream.* Believe that this is Monroe Brook, which drains Monroe but not Eisenhower. Appeared recently avalanched in 1996. This is the largest stream crossed below Gem Pool. Also known as stream 7D.

* *Stream 54.* *884 m. A small stream crossing Ammonoosuc Ravine Trail near Marshfield Station. The lowest stream on the Ammonoosuc Ravine Trail. Spruce-fir, successional status unknown.* Small stream crossing Ammonoosuc Ravine Trail just below the junction with the Ammonoosuc Link (a cutoff to the USFS parking area). Likely logged ~1910. Also known as stream 7C.

* *Stream 55.* *460 m. A small spring-seep surfacing near Rt. 118 immediately above a turnoff onto a dirt road. Kinsman monzonite bedrock and successional northern hardwoods vegetation.* Dirt logging road still exists; it's on the right, the only road encountered off Rt. 118 on descent from Ravine Lodge Rd toward Rt. 112. Stands on the watershed of stream 55 were selectively harvested in 1997-8. Also known as stream 11A.

* *<u>Stream 56.</u> 503 m. A somewhat larger stream crossing under Rt. 118 in a culvert. Fairly deep stream valley. Similar to #55.* No record of cutting on the main watershed since the turn-of-the-century, but small stands on the edge or top may have been cut ~1980. Indications of an old logging road enter off Rt. 118 to the right of the stream when facing upstream. Also known as stream 11B.

<u>Stream 57.</u> 600 m. A small stream crossing under Rt. 118. Similar to #55. Forests above #57 selectively harvested ~1984. No 1973-4 stream chemistry survive.

Table A1.2 GPS locations of stream sampling points; resampled from Vitousek (1975).

	Elev.			NH State Plane (m) 1983 datum		
Stream	MSL (m)	Latitude	Longitude	Northing	Easting	SD (m)
1	1034.9	44.00656	-71.84939	167386.3	285346.5	0.36
2	1029.7	44.00635	-71.84950	167362.4	285337.9	1.38
3	1024.9	44.00582	-71.84977	167303.3	285315.7	0.46
4	1019.2	44.00561	-71.84999	167280.3	285298.3	1.51
5	929.8	44.00315	-71.85383	167008.3	284989.8	1.83
6	909.7	44.00266	-71.85505	166954.0	284891.6	1.13
7	815.0	44.00031	-71.85911	166693.0	284565.3	3.40
8	779.7	43.99968	-71.86095	166624.0	284417.9	1.98
9	730.8	43.99935	-71.86332	166587.1	284227.7	2.35
10	719.0	43.99962	-71.86372	166617.7	284195.8	3.22
13	513.7	43.99775	-71.87540	166411.6	283258.3	1.57
14	770.2	43.99540	-71.81972	166141.6	287724.0	0.64
15	819.7	43.99800	-71.82130	166430.6	287597.1	1.39
16	974.2	44.00893	-71.82795	167645.6	287066.2	3.36
17	979.2	44.00981	-71.82889	167743.7	286991.2	0.40
25	978.5	44.01432	-71.80688	168241.2	288757.4	1.19
26	978.4	44.01405	-71.80647	168210.9	288789.6	1.66
27	957.0	44.01288	-71.80516	168080.8	288894.8	0.59
28	932.4	44.01084	-71.80399	167854.3	288988.5	0.87
29	921.0	44.01005	-71.80390	167767.2	288994.9	1.14
30	896.5	44.01111	-71.80577	167885.1	288845.2	0.88
31	788.3	43.99894	-71.80786	166533.1	288675.3	1.07
32	749.4	43.99663	-71.81298	166276.8	288264.3	3.42
33	750.9	43.99650	-71.81288	166262.7	288272.7	1.53
34	760.5	43.99348	-71.81502	165927.6	288100.4	0.24
35	662.7	43.97883	-71.81887	164300.1	287788.8	2.73
36	648.0	43.97771	-71.81586	164175.1	288030.1	0.84
37	662.1	43.97913	-71.81213	164332.0	288329.6	1.14
38	887.6	44.27337	-71.34715	197065.5	325508.0	1.55
39	875.6	44.27490	-71.34669	197235.5	325544.3	1.07
52	1048.9	44.26755	-71.32687	196424.3	327130.5	0.28
53	982.6	44.26671	-71.33327	196329.3	326619.3	0.55
54	872.9	44.26924	-71.34506	196607.0	325676.9	0.37
55	426.3	43.99494	-71.75124	166082.0	293216.7	16.85
56	478.0	43.99233	-71.75766	165793.2	292701.2	0.83

APPENDIX 2

STREAM SAMPLING DATES, CHEMISTRY, AND MODELED FLOW.

Chapters 1 and 3 describe methods of chemical analyses and streamflow modeling. Appendix 1.1 describes stream location and site descriptions.

Table A2.1: Stream sampling dates

Table A2.2: Nitrate (μg/L)

Table A2.3: Dissolved organic nitrogen (μg/L)

Table A2.4: Ammonium (μg/L)

Table A2.5: Dissolved organic carbon (μg/L)

Table A2.6: Sulfate (μeq/L)

Table A2.7: Calcium (μeq/L)

Table A2.8: Magnesium (μeq/L)

Table A2.9: Sodium (μeq/L)

Table A2.10: Potassium (μeq/L)

Table A2.11: pH

Table A2.12: Modeled streamflow (cm/mo)

Table A2.1: Date of sample collection 1996-7.

	Oct. 1996	Nov. 1996	Jan. 1997	Feb. 1997	Mar. 1997	Apr. 1997	May 1997	Jun. 1997	Jul. 1997	Aug. 1997	Sep. 1997
1A	10/16	11/21	1/14	2/11	3/11	4/17	5/13	6/17	7/15	8/25	9/24
1B	10/16	11/21	1/14	2/11	3/11	4/17	5/13	6/17	7/15	dry	dry
1C	10/16	11/21	1/14	2/11	3/11	4/17	5/13	6/17	7/15	8/25	9/24
2A	10/16	11/21	1/14	2/11	3/11	4/17	5/13	6/17	7/15	8/25	9/24
2B	10/16	11/21	1/14	2/11	3/11	4/17	5/13	6/17	7/15	8/25	9/24
3A	10/16	11/21	1/14	2/11	3/11	4/17	5/13	6/17	7/15	8/25	9/24
3B	10/16	11/21	1/14	2/11	3/11	4/17	5/13	6/17	7/15	8/25	9/24
4A	10/16	11/21	1/14	2/11	3/11	4/17	5/13	6/17	7/15	8/25	9/24
4B	10/16	11/21	1/14	2/11	3/11	4/17	5/13	6/17	7/15	8/25	9/24
4C	10/16	11/21	1/14	2/11	3/11	4/17	5/13	6/17	7/15	8/25	9/24
4D	10/16	11/21	1/14	2/11	3/11	4/17	5/13	6/17	7/15	8/25	9/24
5A	dry	11/21	1/14	2/11	3/11	4/17	5/13	6/17	7/15	dry	dry
5B	10/16	11/21	1/14	2/11	3/11	4/17	5/13	6/17	7/15	8/25	9/24
5C	10/16	11/21	1/14	2/11	3/11	4/17	5/13	6/17	7/15	8/25	9/24
5D	10/16	11/21	1/14	2/11	3/11	4/17	5/13	6/17	7/15	8/25	9/24
6A			1/14	2/13	3/11	4/17	5/15	6/19	7/17	8/25	9/26
6B	10/16	11/21	1/14	2/13	3/11	4/17	5/15	6/19	7/17	8/25	9/26
6C	10/16	11/21	1/14	2/13	3/11	4/17	5/15	6/19	7/17	8/25	9/26
6D				2/13	3/11	4/17	5/15	6/19	7/17	8/25	9/26
7A	10/16	11/21					5/15	6/19	7/17	8/26	9/26
7B	10/16	11/21					5/15	6/19	7/17	8/26	9/26
7C	10/16	11/21					5/15	6/19	7/17	8/26	9/26
7D	10/16						5/15	6/19	7/17	8/26	9/26
7E	10/16						5/15	6/19	7/17	8/26	9/26
8A	10/16	11/21		2/11	3/13	4/15	5/15	6/19	7/17	8/25	9/26
8B	10/16	11/21		2/11	3/13	4/15	5/15	6/19	7/17	8/25	9/26
8C	10/16	11/21		2/11	3/13	4/15	5/15	6/19	7/17	8/25	9/26
8D	10/16	11/21		2/11	3/13	4/15	5/15	6/19	7/17	8/25	9/26
9B							5/15	6/19	7/17	dry	dry
9C							5/15	6/19	7/17	8/25	9/26
9D							5/15	6/19	7/17	dry	dry
10A	10/18	11/21	1/16	2/13	3/13	4/15	5/15	6/19	7/17	8/25	9/26
10B	10/18	11/21	1/16	2/13	3/13	4/15	5/15	6/19	7/17	dry	dry
10C	dry	dry	dry	dry	3/13	4/15	5/15	6/19	7/17	dry	dry
12A							5/13	6/17	7/15	8/26	9/24
12B							5/13	6/17	7/15	8/26	9/24

If blank, stream could not be accessed or was added late in the study.

Table A2.2: Land-use history stream chemistry. Nitrate-N (µg/L).

												Average	
	10/96	11/96	1/97	2/97	3/97	4/97	5/97	6/97	7/97	8/97	9/97	Annual	May-Sept.
1A	0	4	60	66	80	83	66	15	10	7	2	36	20
1B	3	19	25	28	25	31	15	3	4	dry	dry	17	7
1C	0	8	54	52	68	78	62	4	1	1	1	30	14
2A	0	12	29	39	37	40	32	10	2	13	8	20	13
2B	3	8	26	25	31	36	18	4	2	4	0	14	6
3A	5	24	59	*	73	96	72	21	7	3	2	36	21
3B	0	8	30	*	43	50	36	3	3	2	1	18	9
4A	4	55	87	90	104	115	78	29	13	12	10	55	29
4B	142	96	174	213	254	257	186	101	127	79	136	160	126
4C	71	99	188	198	248	333	168	69	98	25	66	142	85
4D	25	64	152	166	190	191	103	58	41	18	22	94	48
5A	dry	0	6	3	8	5	3	1	0	dry	dry	3	2
5B	0	0	9	11	5	9	0	4	4	13	5	5	5
5C	0	4	7	12	6	10	0	4	4	16	6	6	6
5D	0	2	11	24	11	11	0	7	6	22	18	10	11
6A			68	78	92	94	87	17	13	4	1	50	24
6B	2	17	69	75	92	100	66	18	3	4	3	41	19
6C	3	18	51	71	67	88	34	3	7	8	6	32	12
6D				230	293	327	185	110	60	61	59	165	95
7A	72	111					89	77	88	83	78		83
7B	4	5					9	4	9	9	6		7
7C	7	4					8	2	2	3	2		4
7D	179						207	126	160	178	183		171
7E	211						278	209	131	122	178		184
8A				107	140	151	119	49	49	29	30	84	55
8B	5	32		93	116	132	91	41	23	22	9	56	37
8C	17	39		86	103	124	88	36	30	34	18	58	41
8D	121	50		102	103	108	84	48	38	54	79	79	61
9B							26	2	0	dry	dry		9
9C							15	2	2	7	3		6
9D							5	0	1	dry	dry		2
10A	188	117	223	228	291	314	195	195	104	286	228	215	202
10B	71	100	232	257	324	325	247	187	84	dry	dry	203	173
10C	dry	dry	dry	dry	312	373	244	95	46	dry	dry		
12A							154	61	49	32	46		69
12B							102	65	64	32	37		60

If blank, stream could not be accessed or was added late in the study.
* *snow contamination*

Table A2.3: Land-use history stream chemistry. Dissolved organic nitrogen (μg/L).

	10/96	11/96	1/97	2/97	3/97	4/97	5/97	6/97	7/97	8/97	9/97	Average Annual	Average May-Sept.
1A	62	133	55	90	81	89	58	66	66	45	33	71	54
1B	128	96	56	103	59	91	39	40	39	dry	dry	72	39
1C	114	74	85	99	101	79	79	103	138	93	82	95	99
2A	59	85	52	35	75	38	44	40	77	40	39	53	48
2B	57	94	37	40	43	52	75	53	86	47	59	58	64
3A	87	142	105	*	83	68	88	86	148	96	113	102	106
3B	129	106	88	*	73	86	80	59	135	68	61	89	81
4A	37	52	54	55	47	65	89	43	75	25	22	51	51
4B	93	79	74	69	75	56	115	77	54	68	42	73	71
4C	60	76	45	69	83	53	77	71	51	44	97	66	68
4D	57	64	46	71	74	25	84	51	62	51	85	61	66
5A	dry	89	48	35	40	52	66	70	59	dry	dry	57	65
5B	39	52	34	36	27	42	53	34	42	32	62	41	45
5C	62	46	65	34	27	32	55	64	49	26	32	45	45
5D	74	58	54	41	26	47	66	35	34	38	35	46	42
6A			52	71	46	53	82	80	55	52	47	60	63
6B	71	102	67	81	81	58	101	80	69	51	71	76	74
6C	110	108	71	83	80	78	119	126	104	85	94	96	105
6D				75	88	72	103	66	100	13	48	70	66
7A	23	31					62	70	13	38	12		39
7B	67	89					96	102	89	103	69		92
7C	39	90					96	75	52	66	40		66
7D	82						71	31	4	27	21		31
7E	92						61	78	33	37	31		48
8A				62	66	45	66	114	60	38	61	64	68
8B	0	46		44	40	32	46	36	27	9	17	30	27
8C	19	60		39	42	39	62	38	82	8	8	40	40
8D	38	37		35	40	20	44	50	25	0	16	31	27
9B							69	59	46	dry	dry		58
9C							83	76	90	47	58		71
9D							78	93	107	dry	dry		93
10A	141	106	104	119	117	77	81	193	95	32	60	102	92
10B	117	105	119	129	78	77	68	94	81	dry	dry	96	81
10C	dry	dry	dry	dry	37	40	57	101	53	dry	dry		
12A							182	102	111	137	100		126
12B							138	86	72	107	72		95

If blank, stream could not be accessed or was added late in the study.
* snow contamination

Table A2.4: Land-use history stream chemistry. Ammonium-N (μg/L). Some 10/96 and selected other samples may have been contaminated during filtering (c).

	10/96	11/96	1/97	2/97	3/97	4/97	5/97	6/97	7/97	8/97	9/97	Average Annual	Average May-Sept.
1A	7	11	13	7	14	9	8	7	9	9	5	9	8
1B	c	9	12	11	12	11	8	9	11	dry	dry	10	9
1C	8	9	9	5	14	8	9	10	16	17	17	12	14
2A	5	11	11	5	13	11	4	9	9	15	9	10	9
2B	11	12	14	7	9	11	9	10	8	15	13	11	11
3A	9	7	13	*	12	13	8	12	11	24	24	14	16
3B	9	5	14	*	9	12	10	10	6	13	10	10	10
4A	4	8	c	5	6	10	5	9	8	12	10	8	9
4B	9	6	c	6	14	14	3	12	9	9	20	10	11
4C	11	7	c	5	9	10	7	10	16	18	19	11	14
4D	10	9	12	4	13	9	6	6	12	7	22	10	11
5A	dry	5	10	4	15	5	5	9	2	dry	dry	7	5
5B	12	5	9	6	13	10	6	8	11	15	17	10	11
5C	9	12	10	9	8	4	6	9	8	15	14	9	10
5D	16	3	8	7	7	4	10	9	10	19	11	9	12
6A			7	7	13	6	2	3	3	16	2	6	5
6B	12	7	6	7	10	8	2	10	4	3	9	7	6
6C	12	17	c	4	11	6	13	9	8	6	8	9	9
6D				6	14	7	2	3	8	5	11	7	6
7A	c	7					2	0	2	2	6		2
7B	c	7					2	2	5	0	12		4
7C	6	9					3	5	3	0	8		4
7D	5						3	2	3	0	7		3
7E	7						1	12	0	1	18		6
8A				5	4	5	1	3	8	10	7	5	6
8B	c	7		9	9	4	3	2	7	9	12	7	7
8C	8	12		5	4	6	7	4	5	8	7	6	6
8D	9	6		8	3	0	1	2	12	0	10	5	5
9B							6	2	4	dry	dry		4
9C							12	4	10	7	19		10
9D							11	4	20	dry	dry		12
10A	7	9	17	8	3	9	5	9	17	8	17	10	11
10B	11	10	12	3	3	6	6	5	8	dry	dry	7	7
10C	dry	dry	dry	dry	12	3	9	21	12	dry	dry		
12A							23	15	14	4	13		14
12B							13	17	11	1	16		12

If blank, stream could not be accessed or was added late in the study.

* snow contamination

Table A2.5: Land-use history stream chemistry. Dissolved organic carbon (mg/L).

	10/96	11/96	1/97	2/97	3/97	4/97	5/97	6/97	7/97	8/97	9/97	Average Annual	Average May-Sept.
1A	3.2	2.5	1.8	1.6	2.0	2.2	2.8	1.9	3.8	2.2	1.9	2.4	2.5
1B	2.8	1.6	1.3	1.6	1.3	1.5	1.9	1.5	2.3	dry	dry	1.7	1.9
1C	5.6	4.0	3.2	2.9	3.0	3.2	4.9	4.7	9.8	6.0	4.2	4.7	5.9
2A	4.4	2.1	1.7	1.6	1.7	1.9	2.3	2.0	3.0	2.1	1.6	2.2	2.2
2B	2.2	1.9	1.3	1.1	1.3	2.1	2.1	1.9	3.0	2.1	1.6	1.9	2.2
3A	3.2	2.6	2.3	*	2.4	2.7	2.9	2.7	4.8	3.4	2.6	3.0	3.3
3B	3.3	2.5	2.3	*	2.3	2.9	3.1	2.4	5.1	2.9	1.9	2.9	3.1
4A	3.7	1.1	0.9	0.7	1.0	1.4	2.1	1.5	3.7	1.5	1.2	1.7	2.0
4B	2.2	1.5	1.3	1.3	1.2	1.5	1.7	1.6	2.6	2.5	1.9	1.8	2.1
4C	1.9	1.3	1.1	1.2	1.0	1.5	1.4	1.4	2.3	2.1	2.0	1.6	1.9
4D	1.6	1.3	1.2	1.2	1.1	1.4	1.5	1.3	2.1	1.7	1.7	1.5	1.7
5A	dry	1.3	1.2	1.2	1.2	1.6	1.5	1.7	1.8	dry	dry	1.4	1.7
5B	3.2	1.2	1.1	1.1	1.0	1.4	1.2	1.3	1.8	1.4	1.2	1.4	1.4
5C	4.9	1.3	1.1	1.2	1.1	1.5	1.5	1.6	2.5	1.6	1.4	1.8	1.7
5D	4.3	1.2	1.0	1.2	1.3	1.4	1.1	1.4	2.0	1.8	1.4	1.6	1.5
6A			2.1	1.9	2.0	2.5	2.8	4.2	3.6	3.2	2.6	2.8	3.3
6B	2.8	2.7	2.2	1.9	2.2	2.4	2.8	3.2	3.2	2.7	2.3	2.6	2.8
6C	4.7	3.6	3.1	2.6	2.9	3.4	4.8	6.7	6.1	5.0	3.8	4.2	5.3
6D				1.1	1.2	1.4	1.5	1.7	1.6	1.0	1.1	1.3	1.4
7A	1.2	0.8					1.4	2.0	1.2	1.0	0.9		1.3
7B	2.8	2.4					3.0	4.8	3.5	2.7	2.5		3.3
7C	2.2	2.1					3.0	3.4	2.7	2.1	1.5		2.6
7D	1.1						0.9	1.1	0.9	0.8	0.7		0.9
7E	1.1						1.2	2.0	1.3	1.1	0.9		1.3
8A				1.2	1.1	1.3	1.3	3.7	2.0	1.8	1.7	1.8	2.1
8B	1.3	1.0		0.8	0.8	1.0	0.9	1.5	1.3	1.1	1.1	1.1	1.2
8C	1.5	1.2		0.5	0.8	1.0	0.9	1.8	1.4	1.2	1.0	1.1	1.3
8D	1.1	1.0		0.4	0.8	0.9	1.1	1.6	1.3	1.2	1.1	1.0	1.2
9B							1.3	2.1	1.7	dry	dry		1.7
9C							1.7	3.0	2.3	2.1	1.8		2.2
9D							2.8	5.1	3.6	dry	dry		3.8
10A	3.5	3.2	2.1	2.2	2.2	2.7	3.3	7.3	4.1	2.6	1.8	3.2	3.8
10B	2.5	2.3	1.7	1.7	1.6	1.9	1.9	3.2	2.4	dry	dry	2.1	2.5
10C					0.8	1.2	1.2	1.5	1.3	dry	dry		
12A							6.5	4.8	5.8	6.4	4.7		5.7
12B							6.1	4.7	4.9	6.4	4.1		5.2

If blank, stream could not be accessed or was added late in the study.
* snow contamination

Table A2.6: Land-use history stream chemistry. Sulfate (µeq/L). Oct. 1996 samples and 4A, 4B, 4C, and 6C Jan. 1997 (x) should be remeasured.

	10/96	11/96	1/97	2/97	3/97	4/97	5/97	6/97	7/97	8/97	9/97	Average Annual	Average May-Sept.
1A	*90*	76	80	82	75	74	66	86	77	87	83	80	80
1B	*80*	90	71	70	70	69	68	84	76	dry	dry	75	76
1C	*88*	77	78	87	72	70	58	70	58	68	73	73	65
2A	*70*	67	63	81	63	69	59	61	66	67	63	66	63
2B	*85*	94	78	79	96	62	61	74	66	68	78	77	69
3A	*91*	74	69	*	64	65	53	73	66	62	62	68	63
3B	*81*	75	73	*	86	83	60	82	70	76	74	76	72
4A	*83*	50	x	56	55	62	59	46	62	47	44	56	52
4B	*127*	97	x	100	88	75	83	107	96	90	115	98	98
4C	*132*	100	x	94	101	93	87	106	100	100	101	101	99
4D	*116*	87	84	86	80	82	73	93	84	96	100	89	89
5A		71	82	65	89	63	69	75	74	dry	dry	73	73
5B	*107*	76	91	76	80	66	63	82	80	90	91	82	81
5C	*83*	72	64	67	58	63	50	60	61	68	67	65	61
5D	*124*	82	72	79	70	60	61	76	73	84	99	80	78
6A			82	90	80	86	76	62	77	62	75	77	71
6B	*118*	107	102	94	105	96	80	93	102	107	100	100	96
6C	*101*	96	x	95	87	96	68	56	77	83	83	85	73
6D				94	81	82	89	99	105	103	91	93	97
7A	*48*	43					46	40	47	45	42		44
7B	*83*	66					55	57	69	69	65		63
7C	*81*	67					48	58	69	69	67		62
7D	*62*						43	40	47	44	42		43
7E	*62*						37	29	37	37	34		35
8A				74	72	70	66	63	78	47	72	68	65
8B	*72*	70		67	68	65	61	65	72	72	65	68	67
8C	*71*	79		73	66	64	58	62	71	71	65	68	66
8D	*77*	72		64	79	69	61	58	72	67	60	68	63
9B							81	78	80	dry	dry		79
9C							72	77	80	83	77		78
9D							77	89	91	dry	dry		86
10A	*160*	133	125	122	112	104	108	128	133	139	128	127	127
10B	*135*	116	108	104	108	98	127	112	119	dry	dry	114	120
10C					92	92	89	97	104	dry	dry		
12A							59	61	61	56	61		60
12B							52	65	74	69	70		66

If blank, stream could not be accessed or was added late in the study.
* snow contamination

Table A2.7: Land-use history stream chemistry. Calcium (μeq/L).

	10/96	11/96	1/97	2/97	3/97	4/97	5/97	6/97	7/97	8/97	9/97	Average Annual	Average May-Sept.
1A	45	41	51	52	49	43	38	48	51	48	38	46	45
1B	40	51	38	38	35	33	31	40	38	dry	dry	38	37
1C	42	41	43	44	41	36	27	31	34	37	29	37	32
2A	41	34	36	39	39	33	32	32	39	36	27	35	33
2B	54	49	53	54	54	42	43	46	51	52	42	49	47
3A	44	50	42	*	39	36	33	41	39	35	37	40	37
3B	49	48	47	*	43	37	30	45	43	46	52	44	43
4A	35	31	31	33	40	45	39	20	47	33	28	35	33
4B	124	93	93	111	95	75	72	101	93	117	112	99	99
4C	105	90	95	97	97	77	71	96	97	113	92	94	94
4D	68	75	73	72	71	57	55	72	67	73	65	68	66
5A	dry	54	43	43	48	41	37	51	54	dry	dry	46	47
5B	81	45	55	64	55	43	40	66	59	63	67	58	59
5C	63	41	40	45	43	38	33	45	45	48	47	44	44
5D	73	48	42	51	46	36	38	51	48	60	59	50	51
6A			49	49	50	45	37	36	42	43	38	43	39
6B	69	65	64	65	64	57	53	62	66	60	59	62	60
6C	62	53	56	59	54	51	41	42	48	49	45	51	45
6D				64	63	57	61	68	66	64	61	63	64
7A	31	26					29	31	32	34	29		31
7B	33	39					30	40	39	38	35		37
7C	39	31					27	36	35	36	36		34
7D	45						43	37	43	44	26		39
7E	25						26	18	19	13	12		17
8A				74	75	69	61	78	83	66	74	72	72
8B	77	82		75	77	78	75	81	86	67	77	77	77
8C	88	79		80	77	70	65	85	85	66	79	77	76
8D	87	74		73	73	74	58	65	71	71	79	72	69
9B							45	45	53	dry	dry		48
9C							37	30	44	31	53		39
9D							40	48	49	dry	dry		46
10A	106	92	93	91	80	71	66	87	80	64	84	83	76
10B	78	79	77	77	75	70	66	79	75	dry	dry	75	73
10C					59	57	62	79	68				
12A							40	43	48	44	52		45
12B							37	41	50	51	52		46

If blank, stream could not be accessed or was added late in the study.
* snow contamination

Table A2.8: Land-use history stream chemistry. Magnesium (µeq/L).

	10/96	11/96	1/97	2/97	3/97	4/97	5/97	6/97	7/97	8/97	9/97	Average Annual	Average May-Sept.
1A	36	30	33	36	32	28	23	30	29	32	29	31	28
1B	35	35	26	27	24	23	21	26	24	dry	dry	27	24
1C	34	26	28	33	29	25	19	20	21	25	22	26	22
2A	34	24	24	26	25	22	21	19	23	24	21	24	22
2B	25	21	21	21	21	17	17	19	19	21	19	20	19
3A	28	28	24	*	24	23	19	22	21	20	22	23	21
3B	38	32	32	*	29	26	22	27	26	29	38	30	29
4A	25	18	17	19	22	21	19	12	21	18	17	19	17
4B	52	35	34	41	36	28	25	35	30	42	47	37	36
4C	46	36	36	40	40	32	30	38	35	42	42	38	37
4D	27	27	25	28	27	22	20	24	23	24	25	25	23
5A	dry	29	23	24	30	23	20	26	29	dry	dry	25	25
5B	50	30	29	34	30	24	22	34	29	37	40	33	32
5C	42	25	23	27	26	24	20	26	24	26	29	27	25
5D	46	26	24	29	26	21	21	26	23	31	34	28	27
6A			23	23	23	23	17	16	18	18	20	20	18
6B	33	30	30	30	30	28	23	26	26	26	27	28	25
6C	31	25	25	29	27	24	17	17	19	22	23	24	20
6D				26	24	21	23	22	22	22	23	23	23
7A	29	22					22	22	22	24	23		23
7B	18	19					13	16	15	16	17		16
7C	18	16					14	15	15	16	17		16
7D	25						22	18	20	22	14		19
7E	17						16	10	12	9	8		11
8A				18	16	16	12	18	19	19	22	17	18
8B	20	20		17	17	17	16	17	18	15	17	17	16
8C	25	18		19	17	16	12	18	20	17	20	18	17
8D	25	21		20	21	21	16	17	19	19	22	20	19
9B							13	13	14	dry	dry		13
9C							9	7	10	9	12		9
9D							12	13	12	dry	dry		12
10A	31	28	28	30	27	23	22	30	23	20	27	26	24
10B	20	23	23	24	23	22	20	22	19	dry	dry	22	20
10C					17	16	16	15	16	dry	dry		
12A							9	8	9	9	12		9
12B							14	13	16	15	19		15

If blank, stream could not be accessed or was added late in the study.
* snow contamination

Table A2.9: Land-use history stream chemistry. Sodium (μeq/L). Some 10/96 and selected other samples may have been contaminated during filtering.

	10/96	11/96	1/97	2/97	3/97	4/97	5/97	6/97	7/97	8/97	9/97	Average Annual	Average May-Sept.
1A	*28*	23	27	24	24	23	17	21	21	22	17	22	20
1B	*31*	26	21	26	28	25	24	23	18	dry	dry	25	21
1C	*25*	22	22	26	30	23	21	16	19	28	18	23	20
2A	*29*	23	22	26	29	25	23	21	26	30	20	25	24
2B	*41*	46	43	39	47	26	30	34	34	36	34	37	34
3A	*29*	29	26	*	31	26	21	26	21	27	24	26	24
3B	*29*	29	32	*	31	25	19	21	22	24	22	25	21
4A	*29*	26	23	25	28	22	23	24	19	22	22	24	22
4B	*39*	33	26	41	34	32	25	34	27	44	45	35	35
4C	*41*	30	33	38	37	34	32	40	37	41	46	37	39
4D	*37*	33	32	41	36	25	23	30	27	32	40	32	30
5A	*dry*	33	35	35	35	30	30	39	33	dry	dry	34	34
5B	*46*	26	39	46	39	32	32	44	39	41	44	39	40
5C	*39*	39	29	41	39	31	27	31	27	27	41	34	31
5D	*45*	20	26	34	33	23	35	33	27	40	41	33	35
6A			53	36	32	33	23	23	24	22	29	31	24
6B	*41*	53	37	38	36	35	26	35	34	37	39	37	34
6C	*36*	27	40	53	34	31	24	26	27	31	31	33	28
6D				39	38	32	35	41	44	46	45	40	42
7A	*26*	12					17	13	20	21	25		19
7B	*55*	33					22	30	35	38	43		34
7C	*36*	26					26	28	33	32	41		32
7D	*24*						17	15	23	10	19		17
7E	*26*						12	12	15	4	8		10
8A				45	44	36	38	48	55	50	55	46	49
8B	*66*	55		44	44	43	42	45	51	38	51	48	45
8C	*69*	45		45	44	39	40	46	49	33	45	45	42
8D	*51*	54		35	38	42	30	35	37	28	43	39	35
9B							39	41	41	dry	dry		40
9C							33	38	35	26	44		35
9D							35	38	43	dry	dry		39
10A	*67*	53	51	51	59	49	44	45	50	31	55	51	45
10B	*62*	51	44	44	55	44	40	44	44	dry	dry	47	42
10C					48	40	37	32	39	dry	dry		
12A							40	38	42	48	45		42
12B							38	37	65	41	53		47

If blank, stream could not be accessed or was added late in the study.
* snow contamination

Table A2.10: Land-use history stream chemistry. Potassium (µeq/L). Some 10/96 and selected other samples may have been contaminated during filtering (c).

	10/96	11/96	1/97	2/97	3/97	4/97	5/97	6/97	7/97	8/97	9/97	Average Annual	Average May-Sept.
1A	*7.0*	4.6	3.4	4.1	3.8	3.5	2.5	3.3	4.8	1.1	0.7	3.5	2.5
1B	*3.6*	4.1	1.2	3.5	2.7	2.6	1.9	1.6	1.8	dry	dry	2.6	1.8
1C	*3.8*	2.5	3.8	5.0	6.0	4.4	5.5	4.3	4.6	3.7	1.9	4.1	4.0
2A	*10.6*	4.3	4.7	4.4	4.8	4.6	1.6	5.6	4.2	3.4	2.9	4.6	3.5
2B	*4.2*	2.1	8.2	7.5	8.6	3.5	1.0	3.3	1.5	1.6	3.6	4.1	2.2
3A	*2.0*	7.1	3.7	*	3.9	3.5	3.4	3.5	2.2	3.9	1.5	3.6	2.9
3B	*3.4*	2.0	6.2	*	3.9	3.8	0.9	3.0	4.4	2.0	2.6	3.2	2.6
4A	*14.2*	3.5	3.1	4.0	4.8	4.8	3.3	4.2	3.3	1.6	1.2	4.4	2.7
4B	*10.2*	5.9	4.3	6.8	6.8	6.4	4.7	7.9	7.2	8.6	6.6	6.9	7.0
4C	*4.2*	8.4	3.7	4.5	5.3	5.0	3.3	6.0	6.7	2.2	3.7	4.8	4.4
4D	*1.3*	6.3	4.2	5.5	6.4	3.7	3.8	3.5	5.2	2.5	2.4	4.1	3.5
5A	*dry*	2.7	4.0	2.3	3.5	2.0	3.4	3.8	2.8	dry	dry	3.1	3.3
5B	*1.9*	2.8	4.7	7.0	5.0	3.0	4.5	8.4	3.8	4.1	4.9	4.5	5.1
5C	*17.3*	12.3	5.2	7.1	5.8	3.7	4.7	4.6	3.1	3.2	3.4	6.4	3.8
5D	*16.5*	8.4	5.3	5.8	4.8	3.0	4.3	4.7	2.3	3.6	4.0	5.7	3.8
6A			9.9	5.0	5.8	7.1	4.4	5.4	3.4	2.5	4.2	5.3	4.0
6B	*2.4*	9.4	6.0	3.6	5.1	3.8	4.0	3.4	2.1	1.1	2.7	4.0	2.7
6C	*c*	3.2	6.4	11.5	4.7	3.4	3.0	4.5	3.1	3.6	2.8	4.6	3.4
6D				5.3	6.3	7.4	12.0	5.5	4.1	2.4	2.9	5.7	5.4
7A	*8.0*	5.2					7.2	8.6	5.7	7.5	6.3		7.1
7B	*c*	5.7					3.4	3.8	6.3	3.8	5.1		4.5
7C	*c*	3.9					4.6	2.9	2.2	1.7	5.8		3.5
7D	*c*						6.7	5.9	6.3	5.9	9.4		6.9
7E	*5.0*						5.4	4.6	2.9	0.7	7.9		4.3
8A				8.2	8.5	9.1	9.0	8.4	6.6	5.6	5.5	7.6	7.0
8B	*10.7*	15.1		7.4	8.2	8.0	8.6	8.1	8.5	6.3	9.1	9.0	8.1
8C	*10.4*	9.0		8.2	9.0	7.9	12.5	9.0	8.6	4.2	8.1	8.7	8.5
8D	*6.4*	15.7		6.3	6.9	6.4	4.5	5.4	4.9	4.5	6.7	6.8	5.2
9B							3.0	1.7	0.8	dry	dry		1.8
9C							1.7	5.6	1.5	0.3	2.9		2.4
9D							2.1	2.9	2.1	dry	dry		2.4
10A	*5.5*	6.2	6.8	5.2	8.4	5.8	4.9	6.3	4.2	3.3	5.3	5.6	4.8
10B	*8.0*	6.9	4.7	3.6	7.9	5.6	5.0	3.8	1.8	dry	dry	5.3	3.5
10C	*dry*	dry	dry	dry	6.8	5.3	5.2	2.3	2.5	dry	dry		
12A							11.1	8.1	8.2	14.7	10.6		10.5
12B							9.4	8.4	10.7	8.1	11.2		9.6

If blank, stream could not be accessed or was added late in the study.

* snow contamination

Table A2.11: Land-use history stream chemistry. Stream pH.

	10/96	11/96	1/97	2/97	3/97	4/97	5/97	6/97	7/97	8/97	9/97	Average Annual	Average May-Sept.
1A	5.4	5.3	5.4	5.6	5.6	5.5	5.2	5.6	5.7	5.3	6.0	5.5	5.6
1B	5.4	5.0	5.3	5.0	5.1	5.2	5.2	5.4	5.2	dry	dry	5.2	5.2
1C	4.6	4.6	4.8	4.8	4.8	4.9	4.7	4.7	4.5	4.6	4.8	4.7	4.7
2A	5.6	5.3	5.5	5.7	5.6	5.5	5.4	5.7	5.3	5.2	6.0	5.5	5.5
2B	5.7	5.3	5.7	5.6	5.6	5.6	5.6	6.0	5.5	5.4	5.7	5.6	5.6
3A	4.8	4.7	4.9	*	5.0	4.9	5.0	5.0	4.9	4.8	5.2	4.9	5.0
3B	5.2	4.9	5.2	*	5.2	5.0	5.0	5.3	4.8	5.4	5.6	5.2	5.2
4A	5.7	5.6	5.9	6.0	5.9	5.6	5.8	5.7	5.3	5.6	6.2	5.8	5.7
4B	6.2	6.1	6.1	6.5	6.2	5.9	6.4	6.1	5.9	5.7	6.2	6.1	6.1
4C	5.8	5.8	6.2	6.3	6.2	6.0	6.3	6.5	5.8	5.7	6.2	6.1	6.1
4D	6.7	5.6	5.9	5.8	5.8	5.9	6.0	6.1	5.8	5.7	6.0	5.9	5.9
5A	dry	5.7	5.6	5.5	5.9	5.9	5.8	5.9	5.9	dry	dry	5.8	5.9
5B	6.5	6.0	6.1	6.3	6.5	6.1	6.3	6.3	6.2	5.9	6.4	6.2	6.2
5C	6.4	5.6	6.2	6.2	6.0	6.1	6.2	6.3	5.8	6.1	6.4	6.1	6.1
5D	6.2	5.6	5.9	6.0	6.1	5.8	5.8	6.1	5.8	6.0	6.3	6.0	6.0
6A			5.1	5.4	5.4	5.2	5.0	4.8	5.1	5.1	5.7	5.2	5.2
6B	5.8	5.2	5.4	5.5	5.5	5.4	5.5	5.4	5.4	5.4	5.8	5.5	5.5
6C	5.4	4.6	5.0	5.1	5.1	5.0	4.9	4.7	4.8	5.5	5.2	5.0	5.0
6D				5.5	5.4	5.2	5.3	5.4	5.3	5.5	5.9	5.4	5.5
7A	5.9	5.7					6.1	6.1	5.6	6.3	6.3		6.1
7B	5.4	4.8					5.0	5.0	5.1	5.8	5.8		5.4
7C	5.3	4.8					5.0	5.0	5.1	5.4	5.7		5.3
7D	5.9						6.2	5.8	5.8	5.8	6.4		6.0
7E	5.1						4.8	4.7	4.7	5.0	5.4		4.9
8A	dry			6.1	6.1	6.1	6.0	6.2	5.7	5.9	5.9	6.0	5.9
8B	5.9	5.8		6.1	6.1	6.1	6.0	6.0	5.8	5.9	5.9	5.9	5.9
8C	6.0	5.8		6.2	6.1	6.1	6.1	6.2	5.8	6.0	6.2	6.0	6.1
8D	5.9	5.8		5.9	6.1	6.2	6.2	6.2	5.9	5.8	5.9	6.0	6.0
9B							5.5	5.6	5.3	dry	dry		5.5
9C							5.1	5.0	4.9	5.3	5.8		5.2
9D							4.9	4.8	4.8	dry	dry		4.8
10A	5.2	5.2	5.0	5.1	5.0	5.0	4.9	4.8	4.9	5.2	5.5	5.1	5.0
10B	5.5	4.8	5.1	5.2	5.0	5.1	5.1	5.1	5.0	dry	dry	5.1	5.0
10C	dry	dry	dry	dry	4.9	4.9	5.0	4.9	4.8	dry	dry		
12A							4.8	5.0	5.1	5.2	5.6		5.1
12B							4.8	4.9	5.0	5.1	5.5		5.0

Table A2.12: Land-use history streams. Modeled streamflow (cm/month).

	10/96	11/96	12/96	1/97	2/97	3/97	4/97	5/97	6/97	7/97	8/97	9/97	Total
1A	14.2	8.1	7.2	4.5	4.5	4.5	14.6	35.0	4.1	9.4	6.2	4.9	117.3
1B	13.6	8.4	7.6	4.5	4.5	4.5	15.9	31.9	1.4	7.9	5.7	4.3	110.2
1C	14.2	6.9	5.9	4.5	4.5	4.5	12.6	34.0	19.9	13.6	7.9	6.4	135.1
2A	14.0	7.6	6.5	4.5	4.5	4.5	13.7	33.2	12.0	11.5	7.1	5.6	124.7
2B	11.2	8.9	8.7	4.5	4.5	4.7	21.2	15.4	1.0	2.5	3.3	2.6	88.3
3A	12.6	8.7	8.2	4.5	4.5	4.5	18.7	22.3	1.1	5.4	5.2	3.5	99.1
3B	13.3	8.4	7.7	4.5	4.5	4.5	16.4	26.5	1.9	8.7	6.3	4.2	106.9
4A	14.5	6.7	6.0	4.5	4.5	4.5	12.6	32.0	26.2	15.5	8.4	7.0	142.3
4B	14.3	7.3	6.4	4.5	4.5	4.5	14.1	32.1	18.4	12.0	7.4	6.1	131.5
4C	14.0	8.3	7.4	4.5	4.5	4.5	16.0	31.8	5.1	8.6	6.1	4.7	115.4
4D	14.3	7.2	6.2	4.5	4.5	4.5	13.5	32.9	18.9	12.5	7.5	6.2	132.5
5A	12.0	8.8	8.4	4.5	4.5	4.5	19.8	21.2	0.9	3.2	4.3	3.1	95.2
5B	13.0	8.6	8.0	4.5	4.5	4.5	17.9	27.3	1.0	5.4	4.9	3.7	103.2
5C	14.6	7.7	6.5	4.5	4.5	4.5	13.8	35.5	13.0	10.9	6.7	5.7	128.1
5D	13.2	8.5	7.9	4.5	4.5	4.5	17.4	28.5	1.5	5.8	5.0	3.8	105.1
6A	14.0	6.9	5.9	4.5	4.5	4.5	12.1	33.7	19.7	14.3	8.2	6.5	134.9
6B	12.9	8.6	8.0	4.5	4.5	4.5	17.7	23.3	1.3	7.2	5.8	3.8	102.1
6C	13.9	7.1	6.1	4.5	4.5	4.5	12.9	31.9	18.3	13.7	8.2	6.3	131.8
6D	11.5	8.8	8.6	4.5	4.5	4.5	20.7	17.0	0.9	2.4	4.1	2.7	90.2
7A	14.8	5.5	4.9	4.5	4.5	4.5	9.9	29.9	41.6	26.4	10.7	9.1	166.4
7B	14.2	7.2	5.9	4.5	4.5	4.5	12.1	35.8	17.6	14.5	8.5	6.6	136.0
7C	13.8	7.6	6.4	4.5	4.5	4.5	12.2	36.4	10.0	13.4	8.0	5.9	127.1
7D	14.8	4.9	4.6	4.5	4.5	4.5	8.8	27.8	50.0	29.9	11.4	9.9	175.4
7E	15.3	4.6	4.5	4.5	4.5	4.5	6.6	23.8	55.6	51.1	14.1	12.1	201.2
8A	12.4	8.7	8.2	4.5	4.5	4.5	19.0	22.7	1.1	4.6	4.7	3.4	98.2
8B	11.3	8.9	8.7	4.5	4.5	4.6	21.1	14.4	0.9	2.5	3.8	2.5	87.7
8C	11.9	8.8	8.4	4.5	4.5	4.5	19.9	20.7	1.0	3.2	4.2	3.1	94.6
8D	14.1	7.5	6.4	4.5	4.5	4.5	13.9	32.6	14.9	11.6	7.1	5.7	127.4
9B	11.0	8.9	8.7	4.5	4.5	4.6	21.3	15.1	0.9	1.7	3.3	2.5	87.1
9C	11.9	8.8	8.4	4.5	4.5	4.6	19.9	18.5	1.0	3.8	4.3	3.0	93.2
9D	11.6	8.8	8.6	4.5	4.5	4.6	20.5	16.1	0.9	3.4	4.0	2.8	90.4
10A	11.8	8.8	8.5	4.5	4.5	4.5	20.3	17.1	0.9	3.3	4.4	2.9	91.6
10B	12.0	8.8	8.5	4.5	4.5	4.5	20.0	17.5	0.9	4.1	4.7	3.0	92.9
10C	11.3	8.9	8.7	4.5	4.5	4.5	21.0	16.3	0.9	1.9	3.8	2.6	88.8
12A	14.3	6.8	5.5	4.5	4.5	4.5	11.8	34.9	22.4	15.1	8.7	6.9	139.9
12B	14.1	7.3	6.1	4.5	4.5	4.5	12.3	35.7	15.3	13.9	8.2	6.3	132.6

IMAGE EVALUATION TEST TARGET (QA-3)

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