

Long- and Short-Term Changes in Nutrient Availability Following Commercial Sawlog Harvest via Cable Logging

Jennifer D. Knoepp*
Wayne T. Swank
Bruce L. Haines

Introduction

Soil nutrient availability often limits forest productivity, and soils vary considerably in their ability to supply nutrients (Cole 1995). Most southern Appalachian forests are minimally managed, with no fertilizer inputs or routine thinning regime. Nutrient availability is regulated by atmospheric inputs and the internal cycling of nutrients through such processes as forest floor decomposition (Hursh 1928; Alban 1982), soil organic matter mineralization, and the weathering of parent material or primary minerals. Long-term studies in undisturbed forests have shown that soil cation concentrations often decline over time (Binkley et al. 1989; Billett et al. 1990; Knoepp and Swank 1994; Richter et al. 1994). These decreases result from the sequestration of nutrients in aboveground biomass and/or leaching to streams (Johnson et al. 1988; Knoepp and Swank 1994). The sequestration of nutrients has raised questions about the long-term effects of harvesting vegetation and its associated nutrient removal on long-term soil nutrient availability and site productivity.

The responses of soil nutrient pools to forest harvesting vary according to harvest method and forest type. Research has shown that a whole-tree harvest can decrease both total N and exchangeable base cations (Mroz et al. 1985; Waide et al. 1988; Knoepp and Swank 1997). While nutrient declines do not always occur (Hendrickson et al. 1989), the potential for such losses has raised concerns about long-term forest productivity when whole-tree harvesting methods are

* Corresponding author: Coweeta Hydrologic Laboratory, USDA Forest Service, 3160 Coweeta Lab Road, Otto, NC 28763 USA

repeated over several rotations (Federer et al. 1989). In contrast, studies show that a commercial sawlog harvest either increases or has no effect on soil nutrient concentrations (Hendrickson et al. 1989; Kraske and Fernandez 1993), suggesting no negative impact on long-term site productivity (Waide et al. 1988).

Several authors have reported short-lived increases in soil cation concentrations following sawlog harvest (Snyder and Harter 1984; Hendrickson et al. 1989; Kraske and Fernandez 1993). However, forest soil total C and N show varied responses to harvesting depending on site history and treatment (Knoepp and Swank 1997; Johnson et al. 2002b; Johnson et al. 2007). Knoepp and Swank (1997) demonstrated the importance of harvest slash on soil total C and N responses, comparing a whole-tree harvest site with no soil C changes and decreased soil N and a stem-only harvest in which both soil C and N increased.

Forest clearcutting results in changes in nutrient inputs, and nutrient movement within a watershed may occur. While wet deposition (precipitation) is unchanged by forest cutting, dry deposition inputs decrease due to the removal or alteration in aboveground surface area (Boring et al. 1988b). Swank and Henderson (1976) found that dry deposition contributed 19% to 31% of the annual cation input and 7% to 12% of anion input at Coweeta. Total dry deposition was lower by up to 50% in Coweeta than in similar forest communities in the Walker Branch Watershed in Oak Ridge, Tennessee. This difference was attributed to the proximity of coal power plants in Tennessee. Forest structure also has a significant effect on dry deposition input to forests. Studies of cloud and dry deposition to forests and forest edges conducted in the Great Smoky Mountains National Park showed that trees growing on the edge of the forest received about 3 times the input of forest saplings within a gap (Lindberg and Owens 1993). This suggests that inputs may decrease substantially following removal of the forest canopy. Data collected by Potter et al. (1991) in 1985, eight years following site harvest, found that dry-deposition estimates were 2 to 5 times lower than in the Walker Branch Watershed. While some of this difference was attributed to total dry deposition, the effect of the forest canopy was also discussed. In 1985, the WS 7 canopy was estimated to have 85% leaf area index (LAI) of a mature forest, therefore decreasing the rate of dry deposition. Little is known about the recovery of this input source to forests following harvest.

Increases in N availability and increased rates of N transformation processes, mineralization and nitrification, following a forest disturbance or harvest have been reported (Waide et al. 1988; Donaldson and Henderson 1990; Smethurst and Nambiar 1990; O'Connell et al. 2004; Lapointe et al. 2005). It has been suggested that the response is related to the intensity of disturbance, which results in increased soil temperature and water content. Smethurst and Nambiar (1990) found that clearcutting resulted in increased rates of N mineralization in the litter layer 4 months following site disturbance and in the mineral soil after 1 year. This led to increased soil solution concentrations of inorganic N, which leached to depths greater than 30 cm, which is below the major rooting zone. Inorganic N concentrations remained elevated for at least 3 years. Increases in soil N availability were greater in sites where logging slash was left on site. Nutrient movement and losses are related both to the disturbance response and changes in water movement

through the soil profile (Mann et al. 1988; Swank and Vose 1988; Tiedemann et al. 1988; Dahlgren and Driscoll 1994).

Studies showing the effects of harvest on soil nutrients often rely on either short-term data or the chronosequence approach to estimate long-term responses (Mroz et al. 1985; Mann et al. 1988; Hendrickson et al. 1989; Kraske and Fernandez 1993). The objective of our study was to examine the changes in nutrient availability in WS 7 immediately following and for 22 years following harvest. We measured several indices of nutrient availability and nutrient cycling rates including total soil C and N, soil extractable cations, extractable NO_3 and NH_4 , net soil N transformations, soil solution NO_3 concentrations, and nutrient fluxes in slash throughfall. Responses to cutting were measured against both pretreatment soil conditions and by comparison with data collected from an adjacent reference watershed.

Materials and Methods

Site Description

This study was conducted at the USDA Forest Service Coweeta Hydrologic Laboratory in western North Carolina. The clearcut watershed (WS 7) is 59 ha, has slopes of 23% to 81%, and ranges in elevation from 722 to 1077 m. The site was harvested in 1976–1977 using a cable-yarding system. The 12-ha reference watershed (WS 2) has remained untreated since 1927 and ranges in elevation from 709 to 1004 m, with slopes similar to WS 7.

Soil Description

The soils found on the harvested and reference watersheds are similar; fine-loamy to coarse-loamy in texture derived from material weathered from high-grade metamorphosed mica-rich rock or from colluvium. Side-slope soils are in the Fannin and Chandler series and range from 15% to 95% slope. They are well-drained to excessively drained, very deep (solum thickness > 1 m), and > 1.8 m to bedrock. The saprolite layer beneath the solum may be up to 6 m deep (Thomas 1996). Cove or stream side soils were formed in colluvium, have 15% to 50% slope, and are mapped in the Cullasaja-Tuckasegee complex. These soils are very deep and well drained, with solum thickness < 1.5 m and depth to bedrock > 1.8 m (Thomas 1996).

WS 7 sample plots were located on Chandler series (coarse-loamy, micaceous, mesic Typic Dystrochrepts); Fannin series (fine-loamy, micaceous, mesic Typic Hapludults); Cullasaja-Tuckasegee complex (loamy-skeletal or coarse-loamy, mixed, mesic Typic Haplumbrepts); and rock outcrop–Cleveland complex (loamy, mixed, mesic Lithic Dystrochrepts). WS 2 soils include the Fannin series (side-slope) and Cullasaja-Tuckasegee complex (streamside) similar to WS 7. All sample plots were located on the Fannin soil type, which occupies about 60% of the watershed.

Sample Collection and Analysis

WS 7 sample plots (100 m²) were established in 1975 at randomly selected points along 4 transects crossing the watershed (Waide et al. 1988). Pretreatment sampling began in 1975 on 16 plots divided into 2 groups of 8; each group of 8 was sampled alternately every 2 weeks for 17 months. Posttreatment samples were collected on 10 plots, again divided into 2 groups, with alternate groups sampled every 2 weeks. Plots selected for posttreatment soil sampling included both pretreatment plots and sites of intensive vegetation inventory and tree N fixation studies. The biweekly sampling interval continued for 17 months after completion of the harvest. Subsequently, collection frequency decreased, but the alternate-group sampling design continued until 1985. We resampled all 10 plots in 1992, 1993, 1994, 1998, and 2008.

Four 100 m² sample plots on WS 2 were established following the WS 7 harvest in 1977 to serve as long-term reference plots. Plots were divided into 2 groups of 2 and sampled alternately every 2 weeks beginning in July 1977 coinciding with sample collection on WS 7.

Soil Chemical Analyses

Soils in both watersheds were sampled at two depths, 0–10 cm and 10–30 cm. Soils were air-dried and sieved to < 2 mm. Soil C determinations before 1990 were made using the Walkley-Black method (Nelson and Sommers 1982). Regression analysis showed a good correlation between Walkley-Black (WB) and the Leco carbon analyzer with a slope of 1.01 and $r^2 = 0.99$ ($n = 24$). After 1990, soil C was determined using combustion analysis on a Perkin-Elmer 2400 CHN analyzer. Comparisons of total percent C determined with the Leco and the Perkin-Elmer 2400 were conducted using an internal soil standard (4.8% C), certified standard CaHCO₃, and certified standard acetanilide. No significant differences in slope or intercept were detected.

Total N concentrations for all pre-1990 samples were determined using the micro-Kjeldahl method (total Kjeldahl Nitrogen, TKN). Digested solution was analyzed using the cyanurate-salicylate reaction with a segmented flow autoanalyzer (Bremner and Mulvaney 1982). Soil samples collected between 1992 and 2008 were analyzed for total percent N by the combustion method using a Perkin-Elmer 2400 CHN analyzer. Regression analysis between percent N as determined with TKN and combustion was conducted to insure appropriate data comparison. Total percent N by combustion was determined on archived soil samples collected and analyzed for TKN in 1982. Analysis showed a good relationship between the TKN values obtained in 1982 and combustion data with a slope of 1.08 and $r^2 = 0.98$ ($n = 24$). Quality control during soil analysis after 1990 was conducted for total N and C on each set of soil samples.

Soil cation concentrations were determined with the dilute double acid extraction procedure developed by the Soil Testing and Plant Analyses Laboratory, Cooperative Extension Service, Athens, Georgia. Five-gram soil samples were

Table 4.1 Regression analysis output for plasma emission spectroscopy versus atomic absorption spectroscopy analysis of double acid soil extracts.

	Ca ²⁺	Mg ²⁺	K ⁺
Slope	0.95	0.93	0.94
P-value	(< 0.001)	(< 0.001)	(< 0.001)
Intercept	32.2	2.16	2.50
P-value	(0.07)	(0.003)	(0.07)
r ²	0.99	0.99	0.99

Note: Data analyzed were 1993 WS 7 soil cation concentrations (n = 24). Regression values shown with probability of significant value.

extracted with 20 ml of 0.05 M HCl plus 0.05 M H₂SO₄. Pre-1990 soil cation determinations were made using plasma emission spectroscopy (PES) as described by Jones (1977). Although extraction methods did not change in the 1990s, atomic absorption spectroscopy (AAS) was used for base cation concentration determinations. Regression analysis on the 1993 samples compared PES with AAS (table 4.1). AAS data were transformed using these equations before statistical analysis.

Soil Nitrogen Availability Determination

Soil NO₃ and NH₄ concentrations were measured on most soil samples collected for chemical analysis as described above from 1975 to 1980 for WS 7 and 1977 to 1982 for WS 2. We measured soil NO₃ and NH₄ concentrations on all plots in spring 1998 and summer 1999. We conducted laboratory soil incubations (0–10 cm) to determine rates of N mineralization and nitrification during the 1999 growing season in both WS 7 and WS 2 using the methods outlined by Waide et al. (1988). Samples were collected using 15-cm-long, 4.3-cm inside diameter PVC core. Two cores were driven 10 cm into the ground at random locations in each permanent plot. Soil from each core was composited into a single sample for determining NO₃ and NH₄ concentrations and conducting laboratory incubations for potential N mineralization rates. Initial NO₃ and NH₄ concentrations were determined in triplicate for each composited soil sample by extracting 5 g of fresh soil with 20 ml of 2 M KCl. Soil moisture content was determined by drying a 10-g sample overnight at 105°C. We weighed three 10-g subsamples of each composite into 0.94-L jars for laboratory incubation. Jars plus soil were placed in a 25°C incubator (time = 0); soil moisture was adjusted to 33% the following day after soil moisture determinations were complete. Jars were covered with plastic wrap and soil moisture was adjusted weekly as necessary. After 33 days of incubation, 40 ml of 2 M KCl was added to each jar plus soil to extract NO₃ and NH₄. Nitrification rates equal NO₃-N concentration at 33 days minus NO₃-N at time zero. Nitrogen mineralization rates equal NH₄-N + NO₃-N at 33 days minus NH₄-N + NO₃-N at time zero.

Slash Throughfall

We established two, 4 x 4 m plots in each of nine locations (18 plots) in WS 7, which were stratified to proportionally represent precutting forest type for a wood decomposition study. Wood sampling took place 6, 7, and 11 years after cutting (Mattson et al. 1987; see also Mattson and Swank, chapter 7, this volume). Within each 4 x 4 m plot, one nested 2 x 2 m plot was sampled for determinations of fine wood (branches and stems < 5 cm in diameter) inputs following logging and site preparation operations. Plots were selected to represent the range of slash dominated by stems, brush, or mixed slash, all < 5 cm in diameter. Coarse wood was subsampled to estimate surface area and biomass.

We measured slash throughfall by inserting a 15 x 200 cm V-shaped aluminum trough beneath the slash, attached to 19-L polypropylene collection jugs with tygon tubing (figure 4.1). Samples were collected on a storm event basis and used to estimate both slash rainfall interception and nutrient leaching. Samples for each trough were collected weekly and composited monthly by volume yielding weighted composite samples. Solution analyses for NH_4 and NO_3 were conducted on monthly composites as previously described; TKN analyses were conducted on samples composited quarterly during the calendar year; total organic nitrogen (TON) was calculated as $\text{TON} = \text{TKN} - (\text{NO}_3\text{-N} + \text{NH}_4\text{-N})$. We used the annual average throughfall TON concentration and the



Figure 4.1 Throughfall collector beneath slash (one of 12 plots), March 1978. (USDA Forest Service photo)

total throughfall volume to calculate TON fluxes through the slash layer. Data presented are fluxes in $\text{g ha}^{-1} \text{y}^{-1}$.

Soil Solution Collection

After harvest soil solution was collected using both continuous tension porous plate and falling tension 5-cm-diameter porous cup lysimeters. All lysimeters were washed prior to field installation using 1 N HCl (Grover and Lamborn 1970) followed by rinsing with deionized water until electrical conductivity of the leachate equaled the deionized water. Porous plate lysimeters were constructed from 15-cm-diameter ceramic plates (Pacific Lysimeter, Seattle, Washington) fitted with a fiberglass resin backing and plastic drain tubes. Solutions drained from the lysimeters to 25-L vented plastic carboys. Tension porous plate lysimeters were inserted 30 cm below the soil surface. Drain elevations were 100 cm below the surface of the ceramic plates, and were filled with water providing a 100-cm hanging water column, approximately -0.01 MPa. Porous cup lysimeters were installed in pairs at 30 and 100 cm beneath the litter-soil interface. Porous cup lysimeters were evacuated weekly with a hand pump to -0.03 MPa.

Tension porous plates were placed at 30 cm depth in all 10 plots on WS 7 and 4 on WS 2. Samples were collected once a month for analysis during 1978 and 1979. Falling tension porous cup lysimeters were installed at 30 cm and 100 cm and sampled monthly 1980 through 1984. Porous cup lysimeters were installed on a subset of the sample plots with similar vegetation and soil types, four on WS 7 and three on WS 2. Soil solution analyses were conducted as described for stream water by Swank et al. (see chapter 3, this volume).

Statistical Analysis

Statistical analyses for differences between pretreatment and posttreatment years in soil extractable Ca, Mg, and K concentrations for each watershed were tested through analysis of variance with the Mixed Procedure of SAS (SAS 2000). We used WS(Plot) in the random statement and determined significant differences ($P \leq 0.10$) among means using Tukey adjusted LSMeans. All WS 7 pretreatment soil-chemistry data from 1975 and 1976 were compared with annual posttreatment means. We compared WS 7 and WS 2 soil chemistry and soil solution chemistry year-by-year using the 6 plots on WS 7 with soils similar to those on WS 2. We conducted analysis of variance with the GLM Procedure of SAS (SAS 2000) using WS(Plot) as the error term, with significant differences ($P \leq 0.10$) reported. Plot soil series determination was based on the Coweeta soil map (Thomas 1996) with plot level ground truthing.

Differences between WS 7 and WS 2 soil NO_3 and NH_4 concentrations ($t = 0$) for all sample years were determined as described above using annual plot means from plots in similar soil types on WS 7 and WS 2. Pre-1999 data for laboratory incubation N mineralization and nitrification rate measurements were not available;

Table 4.2 Precipitation inputs and net throughfall inputs for WS 7 postharvest logging slash and WS 2.

Year	Site/Material	NO ₃ -N	NH ₄ -N	PO ₄	SO ₄	Ca	K
Precipitation inputs (g ha⁻¹ yr⁻¹)							
1977-1979		2680	1569	251	28674	3950	1928
Net throughfall (g ha⁻¹ yr⁻¹)							
1978	WS 7 postharvest slash	1251	694	-3843	-25610	-9701	-35929
1985-1988*	WS 2 reference forest	1351	-826	-325	-2928	-7240	-21489

Note: Net throughfall equals precipitation input minus throughfall. Values shown are annual means for the time period. Net throughfall values < 0 indicate nutrient leaching from canopy and values > 0 indicate canopy uptake.
*Throughfall data collected during the Integrated Forest Study (Swank et al. 1992).

therefore, statistical comparisons between past and recent data were not possible. Previously published mean values and standard errors are presented (Waide et al. 1988).

Results

Nutrient Input Response

Post-harvest total bulk precipitation of nutrients was typical compared to years before forest harvest (table 4.2). However, the removal of the overstory changed nutrient input to the site. For example, slash throughfall chemistry on WS 7 varied substantially in NH₄, SO₄, and PO₄ compared to throughfall concentrations in the adjacent reference watershed, WS 2, measured during the Integrated Forest Study funded by the Electric Power Research Institute (Swank et al. 1992) resulting in differences in net throughfall (table 4.2).

Soil Carbon and Nitrogen Response

Surface soil (0-10 cm) C increased significantly after forest harvest. C content in surface soils averaged 32.1 Mg/ha during the 2 sampling years prior to harvest and increased to an average of 50 Mg/ha for 3 years after forest cutting (figure 4.2A). Surface soil C remained greater, although differences were significant in only 2 of the next 28 years of sampling. Responses of subsurface (10-30 cm) soils C differed (figure 4.2B). Total C did not respond in the first 3 years following cutting, however, beginning in 1980, subsurface C was significantly lower compared to pretreatment levels in 5 of the 13 posttreatment sampling years.

Total N in the surface soil (0-10 cm) increased significantly compared to preharvest N content for 3 years following forest harvest (figure 4.3A). Total N content increased from 1.4 Mg/ha preharvest mean to 2.2 Mg/ha in 1977 through 1979. After this initial response, total N did not differ significantly from preharvest levels

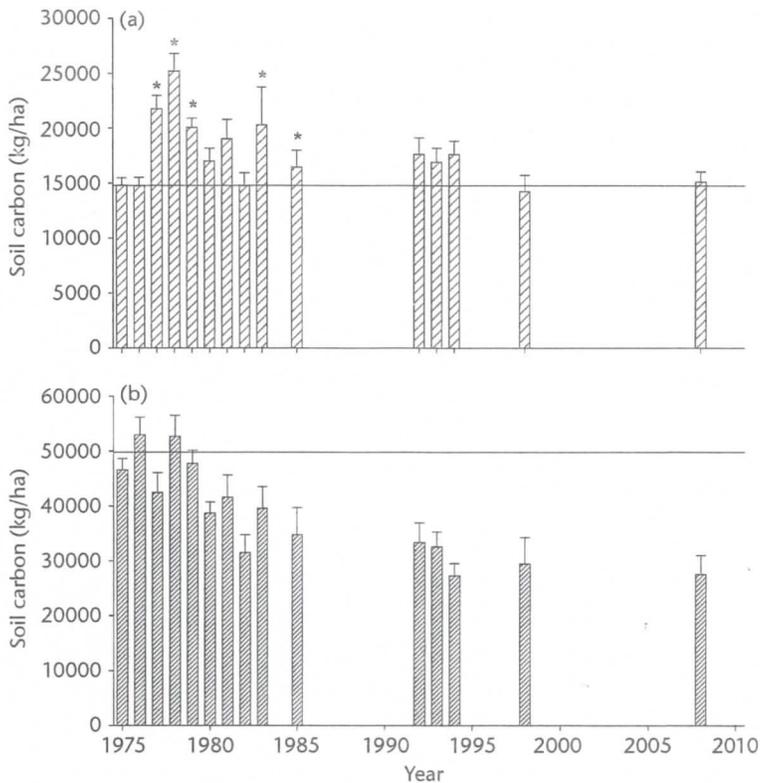


Figure 4.2 Soil carbon (C) content (kg/ha) for (A) 0–10 cm and (B) 10–30 cm collected between 1975 and 1998 on harvested WS 7. Annual means were calculated using annual mean values for each plot (preharvest, $n = 16$; postharvest $n = 10$). The line represents the mean C content of preharvest soils. Error bars represent standard errors of the mean. Bars marked with ‘*’ have a significant difference between pre- and postharvest C content ($P \leq 0.10$).

for any of the next 8 sample collections over 19 years. Subsoil total N content averaged 3.1 Mg/ha preharvest and did not change immediately following cutting. However, subsoil N decreased to < 2.0 Mg/ha by the final sample collection in 2008 (figure 4.3B).

Comparing the seven plots on WS 7 with soils similar to WS 2, the reference watershed (postharvest collections only), suggests a different pattern of total soil C and N response over time (figs. 4.4 and 4.5). For example, surface soil total C increases were significant for 5 years following harvest and were greater in two of the next seven sample collections (figure 4.4A). On the other hand, comparing WS 7 with WS 2 suggests no significant change in total C in the subsoil layer. Comparing WS 7 and WS 2 suggests that both watersheds experienced a long-term decline in subsurface soil C during the postharvest sampling years (figure 4.4B). A comparison of WS 7 and WS 2 total N content showed a significant increase for only one year after harvest (1979, figure 4.5A) in the surface and subsurface soil layer (figure 4.5B).

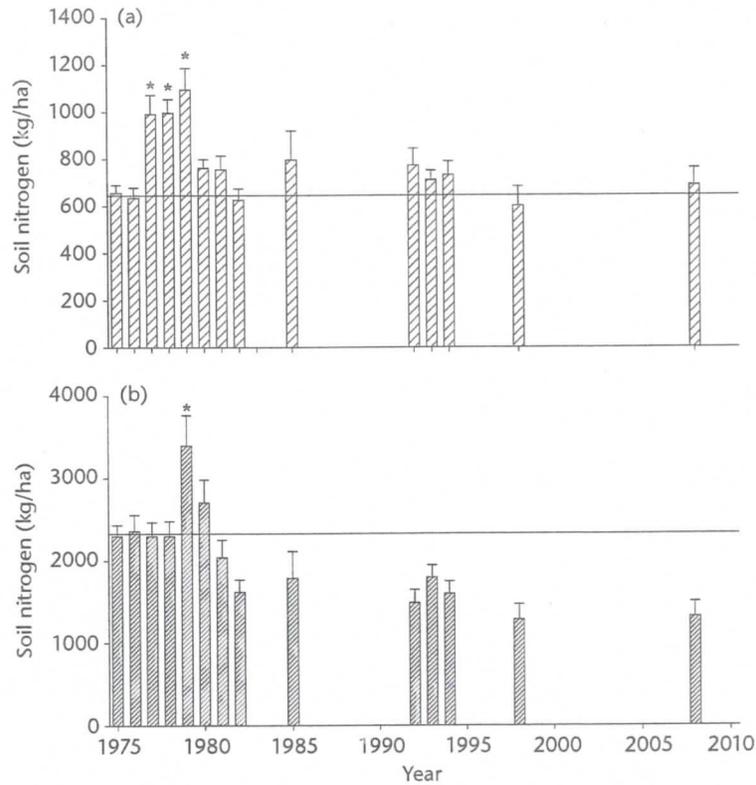


Figure 4.3 Soil nitrogen (N) content (kg/ha) for (A) 0–10 cm and (B) 10–30 cm collected between 1975 and 1998 on harvested WS 7. Annual means were calculated using annual mean values for each plot (preharvest, $n = 16$; postharvest $n = 10$). The line represents the mean N content of preharvest soils. Error bars represent standard errors of the mean. Bars marked with ‘*’ have a significant difference between pre- and postharvest N content ($P \leq 0.10$).

Soil Cation Responses

We measured a significant increase in surface soil Ca, Mg, and K content for 3 years (1977–1979) following harvest compared to preharvest cation content (figures 4.6A, 4.7A, 4.8A). Increases averaged over 100% for each cation. Following the initial cation increase, responses during the years 1980–2008 differed for each cation. During the post-1980 years, Ca and K content did not differ from preharvest levels. Mg content postharvest was significantly greater than it was pretreatment in 1992 and 2008.

Subsurface soils responses also differed among the three base cations. There were no significant differences between pre- and postharvest subsoil Ca content (figure 4.6B). Subsoil Mg responses were similar to surface soils, 2 of 3 postharvest years and 2 of 4 years in the 1990s had significantly greater Mg content compared to preharvest values (figure 4.7B). Soil K content response in the subsurface soil layer

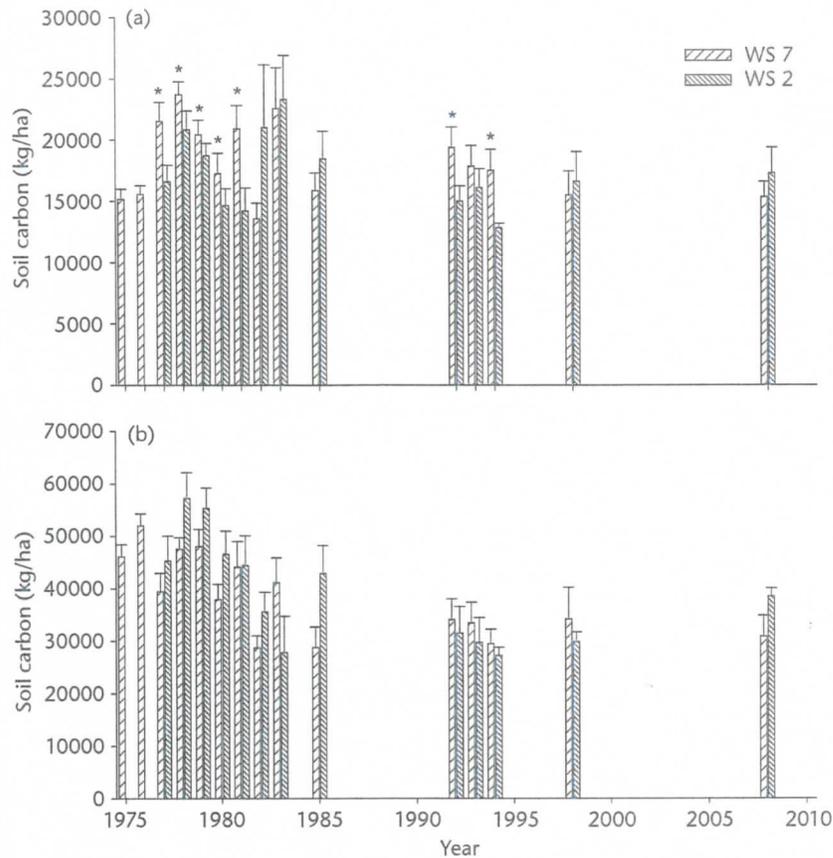


Figure 4.4 Soil C content (kg/ha) for (A) 0–10 cm and (B) 10–30 cm collected between 1977 and 1998 on harvested WS 7 and reference WS 2. Annual means were calculated using annual mean values for each plot (WS 7 $n = 6$; WS 2 $n = 4$). Error bars represent standard errors of the mean. Bars marked with ‘*’ have a significant difference between WS 7 and WS 2 ($P \leq 0.10$).

differed from K surface soils and other cations in subsurface soils. Post-harvest K content was significantly greater than preharvest content in 10 out of 11 postharvest sample collection years (figure 4.8B).

Similar to soil C and N responses, comparing plots on harvested and reference watershed with the same soil series resulted in different conclusions regarding soil cation responses to harvest. Surface soil Ca content showed no significant response to harvest, except in 1992 (figure 4.9). However, this analysis suggests that the Ca content of the subsurface soil 10–30 cm soil layer was significantly lower in WS 7 compared to WS 2 in 3 of the first 4 postharvest years. Soil Mg content responses also differed using this comparison (figure 4.10). In 1979, 3 years after harvest, surface and subsurface soil Mg content was greater in WS 2 compared to WS 7. Soil Mg content did not differ significantly between the two watersheds in any other

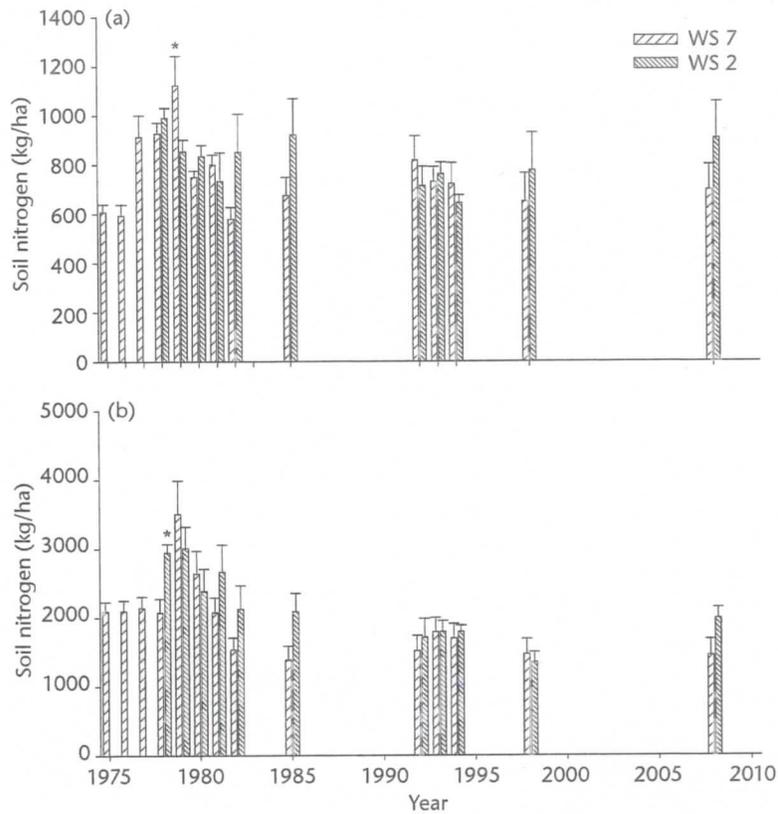


Figure 4.5 Soil N content (kg/ha) for (A) 0–10 cm and (B) 10–30 cm collected between 1977 and 1998 on harvested WS 7 and reference WS 2. Annual means were calculated using annual mean values for each plot (WS 7 $n = 6$; WS 2 $n = 4$). Error bars represent standard errors of the mean. Bars marked with '*' have a significant difference between WS 7 and WS 2 ($P \leq 0.10$).

sample years. Soil K responses were similar to the other cations (figure 4.11). There were few significant differences between WS 7 and WS 2, and in those years K content was greater in WS 2.

Nitrogen Availability

Waide et al. (1988) found significant increases in extractable soil NH_4 concentrations following harvest from 1977–1980. NH_4 concentrations averaged 8.9 mg N/kg in WS 7 compared to 3.1 mg N/kg in WS 2 (table 4.3). Differences in NO_3 concentrations between WS 7 and WS 2 were similar in magnitude, but due to high variability were not significant. The NH_4 concentrations in soils collected in 1998 and 1999 were 50% greater in WS 7 compared to WS 2, while NO_3 was 10 times greater; neither was significantly different (table 4.3).

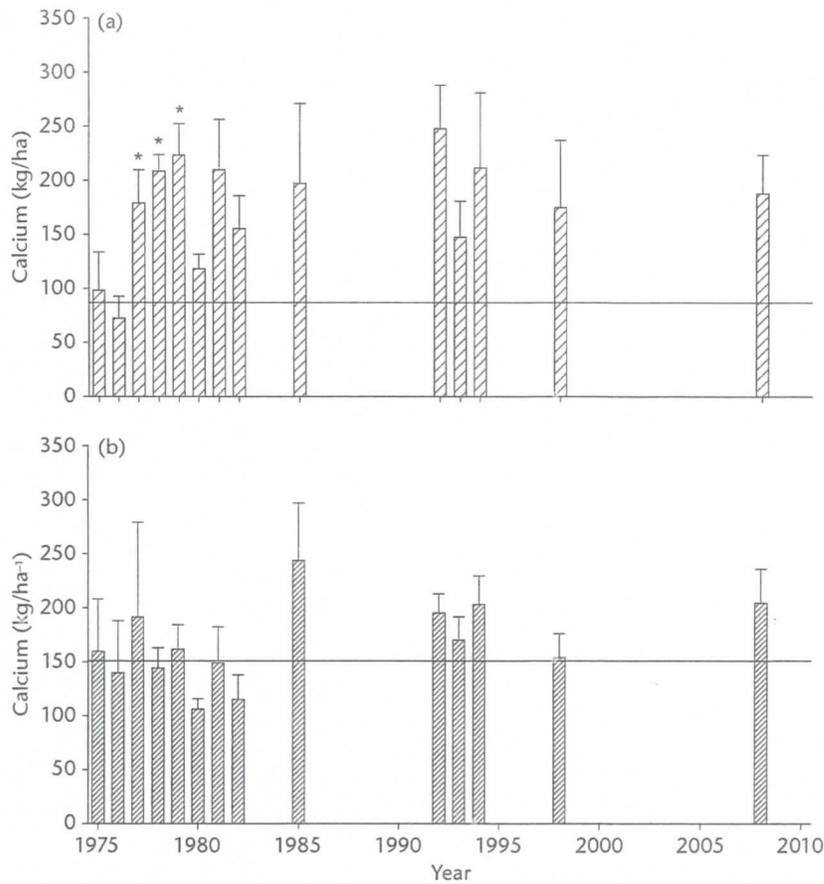


Figure 4.6 Soil calcium (Ca) content (kg/ha) for (A) 0–10 cm and (B) 10–30 cm collected between 1975 and 1998 on harvested WS 7. Annual means were calculated using annual mean values for each plot (preharvest, $n = 16$; postharvest $n = 10$). The line represents the mean Ca content of preharvest soils. Error bars represent standard errors of the mean. Bars marked with ‘*’ have a significant difference between pre- and postharvest Ca content ($P \leq 0.10$).

Potential net N mineralization and nitrification was measured using laboratory incubations at constant temperature and soil moisture during the years 1979–1982 and again in 1999. Early data suggested a slight increase in N mineralization that could have contributed to the increase in available soil N concentrations. There was also a significant increase in nitrification rates at all soil layers tested, Oa, 0–10 cm, and 10–30 cm soil depths. Again, Waide et al. (1988) suggested that the increase in nitrification potential was consistent with the increase in soil extractable NO_3^- . In 1999 laboratory incubations, net N mineralization and nitrification were 3.4 and 20 times greater in WS 7 compared to WS 2, respectively. However, due to high variability these differences were not statistically significant (table 4.3).

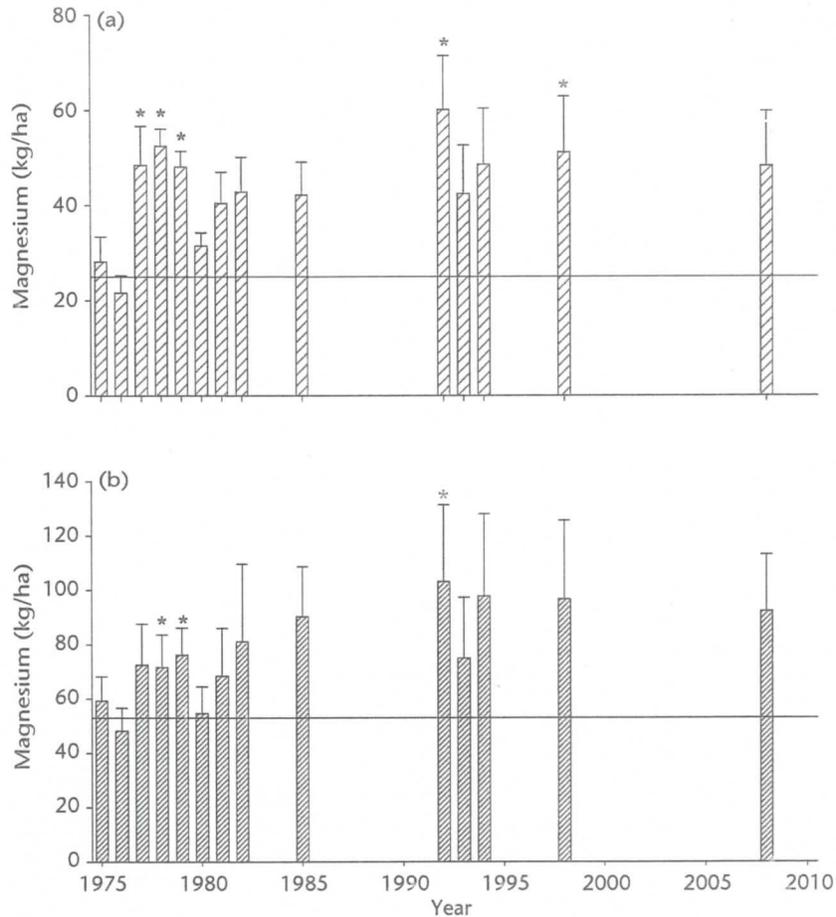


Figure 4.7 Soil magnesium (Mg) content (kg/ha) for (A) 0–10 cm and (B) 10–30 cm collected between 1975 and 1998 on harvested WS 7. Annual means were calculated using annual mean values for each plot (preharvest, $n = 16$; postharvest $n = 10$). The line represents the mean Mg content of preharvest soils. Error bars represent standard errors of the mean. Bars marked with ‘*’ have a significant difference between pre- and postharvest Mg content ($P \leq 0.10$).

Soil Solution Responses

Postharvest annual means of soil solution collected using tension lysimeters at 30 cm showed that overall WS 7 had significantly greater NO_3 and Ca in solution compared to WS 2 (figure 4.12). In yearly comparisons of WS 7 and WS 2 soil solution concentrations of NO_3 in 1984, Ca in 1983 and 1984, and K in 1984 were significantly greater in WS 7. In 1983, Ca concentrations in soil solutions collected at 100 cm were also significantly greater in WS 7 compared to WS 2 (figure 4.13).

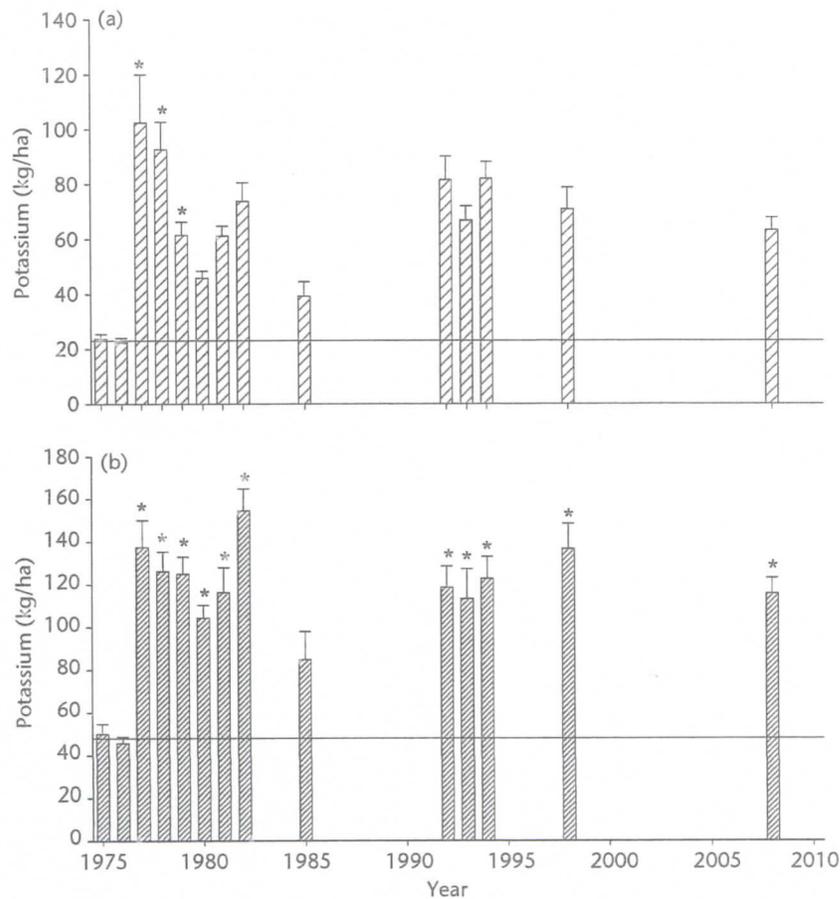


Figure 4.8 Soil potassium (K) content (kg/ha) for (A) 0–10 cm and (B) 10–30 cm collected between 1975 and 1998 on harvested WS 7. Annual means were calculated using annual mean values for each plot (preharvest, $n = 16$; postharvest $n = 10$). The line represents the mean K content of preharvest soils. Error bars represent standard errors of the mean. Bars marked with '*' have a significant difference between pre- and postharvest K content ($P \leq 0.10$).

Discussion

Nutrient Inputs

Patterns of throughfall collected beneath logging slash compared to undisturbed forest are indicative of the impacts of forest harvest on nutrient inputs via dry deposition and impacts of canopy processes on nutrient fluxes (table 4.2). The net input of NH_4 to the forest floor was lower in the clearcut forest compared to the reference; the slash canopy immobilized N, while the N was removed from the reference canopy. Inputs of PO_4 and SO_4 to the forest floor were also altered, and while both were leached from slash and canopy materials, the leaching of PO_4 and SO_4 was

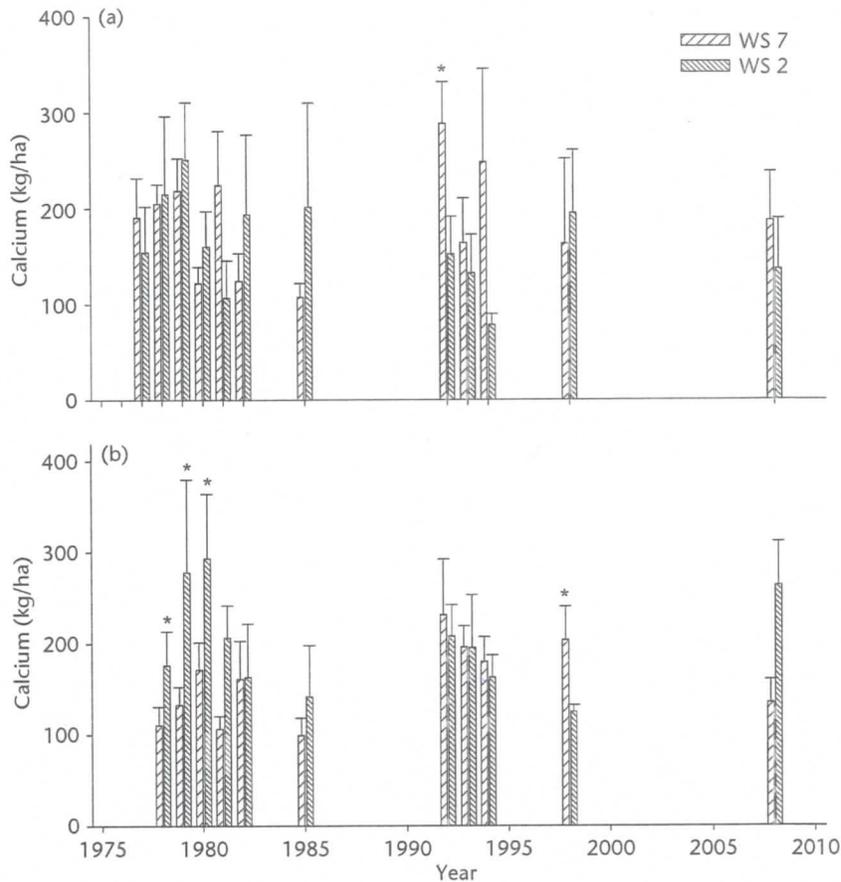


Figure 4.9 Soil Ca content (kg/ha) for (A) 0–10 cm and (B) 10–30 cm collected between 1977 and 1998 on harvested WS 7 and reference WS 2. Annual means were calculated using annual mean values for each plot (WS 7 $n = 6$; WS 2 $n = 4$). Error bars represent standard errors of the mean. Bars marked with ‘*’ have a significant difference between WS 7 and WS 2 ($P \leq 0.10$).

greater from the slash material. As the forest regrows and the canopy structure and LAI return to preharvest conditions, these values should recover. A detailed study of atmospheric deposition and foliage leaching was conducted on WS 7 when the regenerating forest was 8 years old (Potter et al. 1991; Potter 1992). They collected rainfall and dry deposition as well as throughfall and stemflow in an area dominated by chestnut oak (*Quercus prinus*). These water fluxes for the regenerating stand were similar in magnitude to those measured concurrently in the adjacent reference hardwood forest on WS 2 (Swank et al. 1992).

Forests of the southern Appalachians are rarely fertilized and so nutrient availability is regulated by inputs to the system from the atmosphere and soils plus the

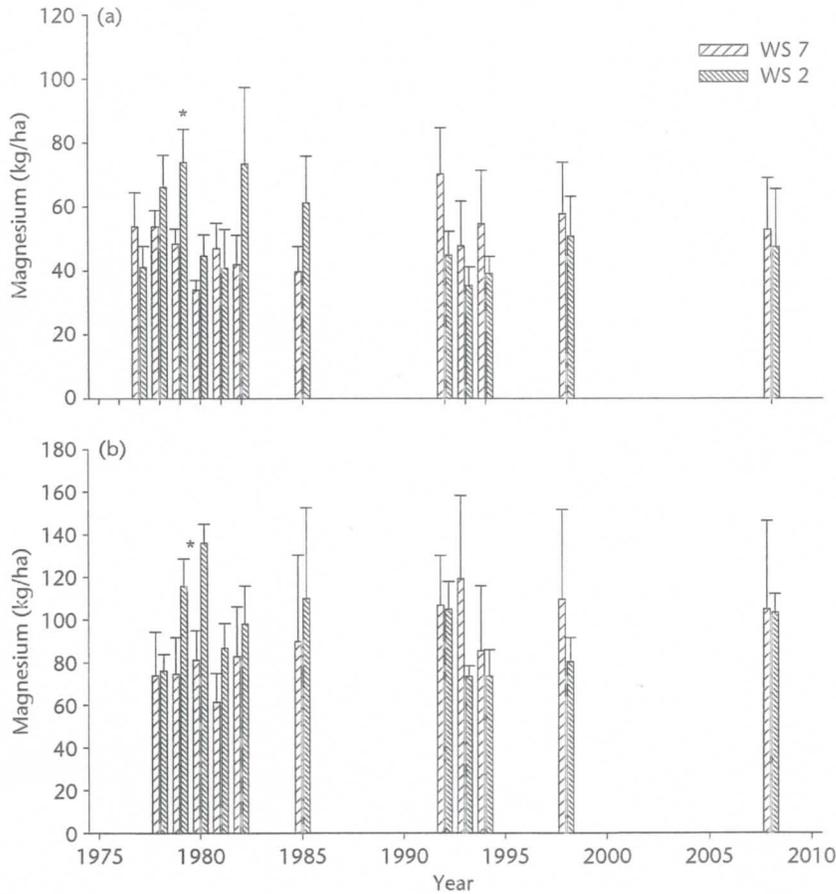


Figure 4.10. Soil Mg content (kg/ha) for (A) 0–10 cm and (B) 10–30 cm collected between 1977 and 1998 on harvested WS 7 and reference WS 2. Annual means were calculated using annual mean values for each plot (WS 7 $n = 6$; WS 2 $n = 4$). Error bars represent standard errors of the mean. Bars marked with ‘*’ have a significant difference between WS 7 and WS 2 ($P < 0.10$).

cycling of nutrients input through litterfall and subsequent decomposition (Hursh 1928; Alban 1982). Research examining inputs by throughfall and litterfall by Eaton et al. (1973) and later by Gallardo et al. (1998) suggested that C, N, and P associated with organic compounds move as litterfall from the canopy to the forest floor and then to the soil through decomposition; whereas ions such as K move rapidly from the canopy to the soil by throughfall and stemflow. Patterns of canopy retention or leaching differ by nutrient both seasonally and spatially (Eaton et al. 1973; Andersson 1991; Knoepp et al. 2008; Knoepp et al. 2011).

74 Long-Term Response of a Forest Watershed Ecosystem

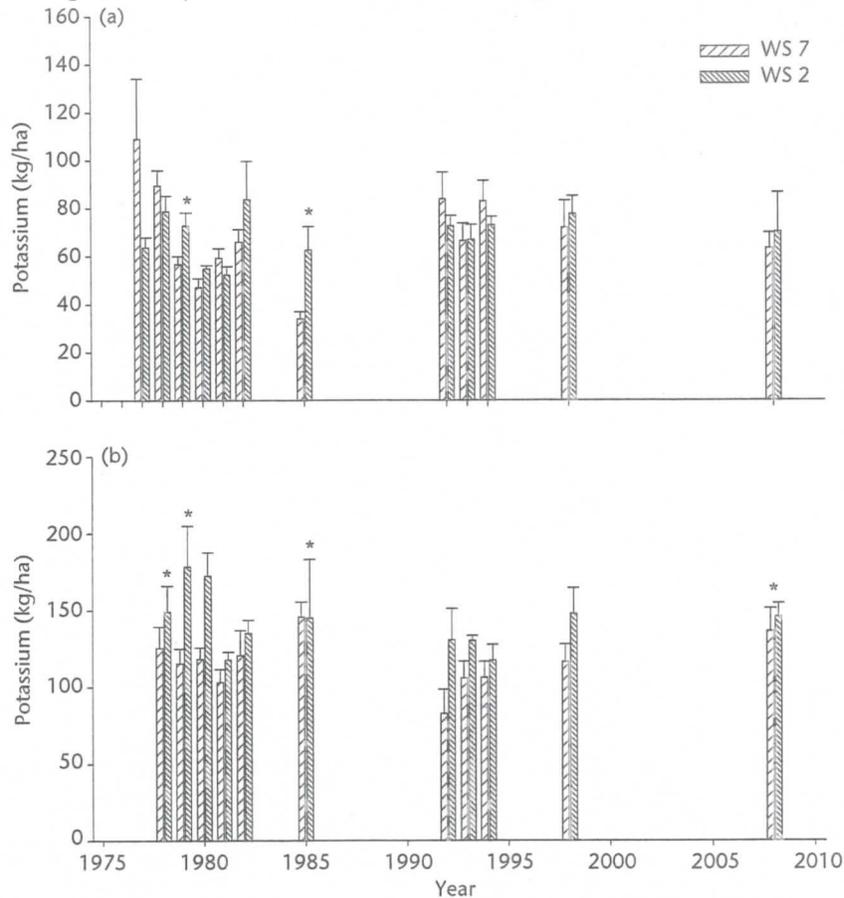


Figure 4.11. Soil K content (kg/ha) for (A) 0–10 cm and (B) 10–30 cm collected between 1977 and 1998 on harvested WS 7 and reference WS 2. Annual means were calculated using annual mean values for each plot (WS 7 n = 6; WS 2 n = 4). Error bars represent standard errors of the mean. Bars marked with ‘*’ have a significant difference between WS 7 and WS 2 ($P \leq 0.10$).

Nutrient Pools

We measured an increase in total soil C and N in the sawlog-only harvest in this study; however, soils in a whole-tree harvest site had no C response and a significant decrease in N following cutting (Knoepp and Swank 1997). Many research studies attribute the initial soil nutrient responses to inputs of slash or logging residue to the forest floor. For example, Shammass et al. (2003) examined the impacts of harvest slash on nutrient cycling in Eucalyptus plantations in Australia. They estimated that the total N in the logging residue equaled 11% of the total N in the surface soils, but up to 100% of exchangeable cations. The largest pool of nutrients was in the leaves, which contributed 30% of the total mass and 75% of the total N. In a *Pinus radiata* plantation in Australia, Baker et al. (1989) examined

Table 4.3 Soil N transformation rates and KCl extractable N concentrations for the surface soils (0–10 cm) on WS 7 and WS 2.

A.	1979–1982		1999		
	WS 2	WS 7	WS 2	WS 7	WS 7 (n = 6)
Mineralization (mg N kg ⁻¹ 33d ⁻¹)	17.8 (3.2)	21.7 (3.1)	5.8 (1.4)	25.5 (6.5)	17.0 (5.7)
Nitrification (mg N kg ⁻¹ 33d ⁻¹)	4.5 (1.7)	15.4 (3.1)	0.3 (0.2)	6.3 (1.9)	2.6 (1.4)
B.	1977–1980		1998–1999		
	WS 2	WS 7	WS 2	WS 7	WS 7 (n = 6)
NH ₄ -N (mg/kg)	3.1 (0.3)	8.9 (1.2)	1.8 (0.41)	2.7 (0.41)	2.04 (0.28)
NO ₃ -N (mg/kg)	1.6 (0.4)	3.2 (0.6)	0.01 (<0.01)	1.0 (0.47)	0.17 (0.10)

Note: (A) Data presented for 1979–1982 are from Waide et al. (1988). The 1999 data are means of 2 laboratory incubations (25°C) conducted in May and July (standard errors in parentheses). Mineralization and nitrification rates presented are for all WS 7 plots (comparable to Waide et al. 1988) as well as plots (n = 6) with soils similar to those on WS 2. (B) Soil ammonium and nitrate nitrogen concentrations at collection (prior to incubation) for incubations conducted 1977–1980 as well as in situ incubations conducted 1998–1999. Concentrations presented are for all WS 7 plots (comparable to Waide et al. 1988) as well as plots (n = 6) with soils similar to those on WS 2.

patterns of nutrient release from logging slash. They found that nutrient leaching began after 3 months of decomposition and that after 12 months, 16% of the total N had been released, while 19% of the Ca and 92% of K had been leached from the slash material.

The cable-yarding sawlog harvest method used in this study, removed only about 10% of the total aboveground biomass (Boring et al. 1988a), leaving 121 Mg/ha of logging residue (Mattson et al. 1987) fairly well distributed across the site. Mattson et al. (1987) estimated a composition of 91.2 Mg/ha of coarse woody material and 30.3 Mg/ha fine wood. The branches, boles, and roots represent a long-lasting source of nutrients and organic matter. Inventory of woody residue 6 years after cutting found 58% of the coarse wood and 25% of the fine wood still remaining on the site (Mattson et al. 1987).

Leaves and small twigs, which decompose rapidly, provide another immediate source of exchangeable cations and soil organic matter (Abbott and Crossley 1982) and may represent an immediate source of cations to the soil beneath the harvest slash (table 4.2). Decomposition of this material initially immobilizes some nutrients, such as Ca, but subsequently releases them as it continues to decompose (Abbott and Crossley 1982; Fahey et al. 1988). This process may also limit the loss of dissolved organic material to streams that occurs following harvest (Mann et al. 1988; Swank et al. 1988; Tiedemann et al. 1988; Dahlgren and Driscoll 1994).

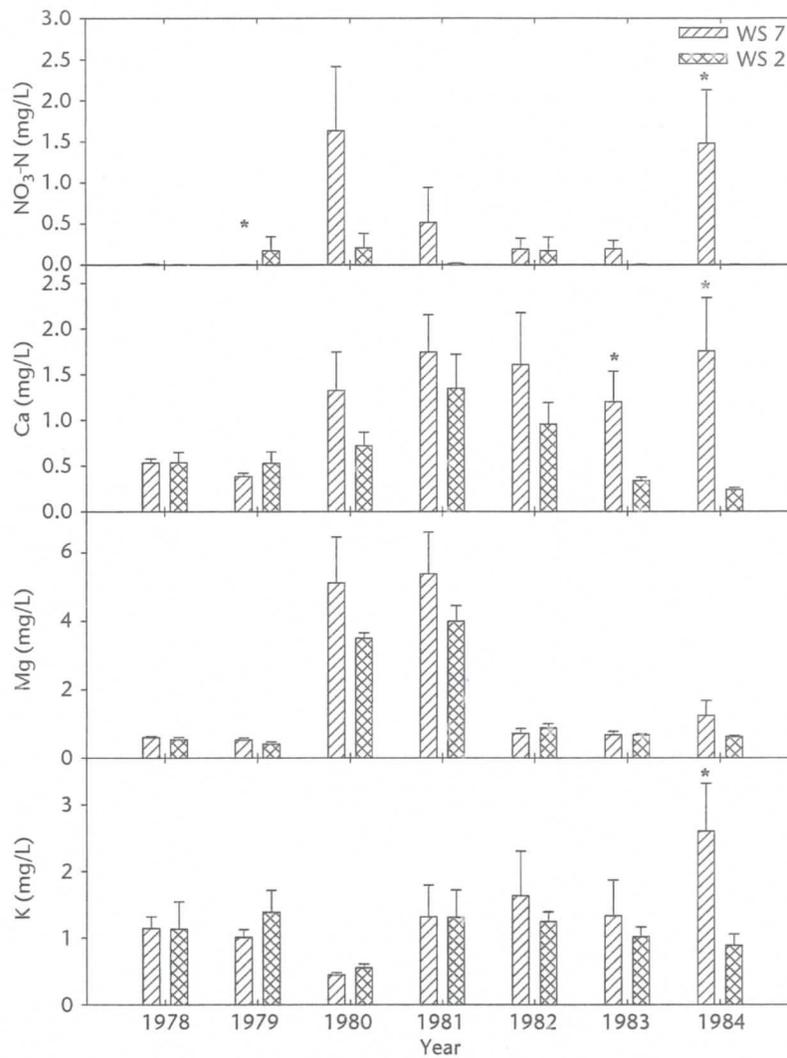


Figure 4.12 Soil solution concentrations (mg/L) of $\text{NO}_3\text{-N}$, Ca, Mg, and K, collected using continuous tension porous plate (1978 and 1979) and falling tension porous cup lysimeters (1980–1984) placed at a soil depth of 30 cm. Values represent means of annual plot means for WS 7 (5 plots) and WS 2 (4 plots), bars represent the standard error of the mean. Bars marked with '*' have a significant difference between WS 7 and WS 2 ($P \leq 0.10$).

Long-term Soil C and N Response

In before and after comparisons, our data show that after harvest there was an initial increase in surface soil total C and N followed by a long period without further change. However, comparison of harvested WS 7 with reference WS 2 shows a decline in total C and N over the 30+-year sampling period for both watersheds.

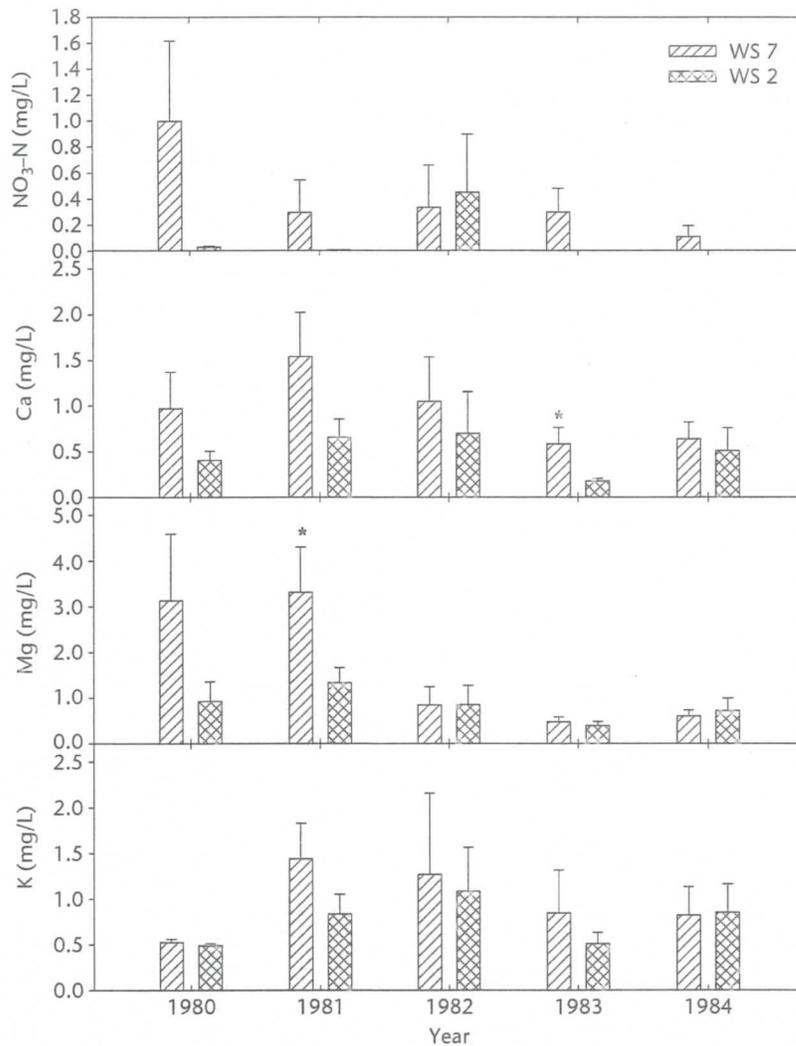


Figure 4.13 Soil solution concentrations (mg/L) of NO₃-N, Ca, Mg, and K, collected using falling tension porous cup lysimeters (1980–1984) placed at a soil depth of 100 cm. Values represent means of annual plot means for WS 7 (5 plots) and WS 2 (4 plots), bars represent the standard error of the mean. Bars marked with '*' have a significant difference between WS 7 and WS 2 ($P \leq 0.10$).

Studies of long-term changes in forest soil total C and N show varied responses depending on site history and treatment (Knoepp and Swank 1997; Johnson et al. 2002b; Johnson et al. 2007). Both Johnson et al. (2002b) and Knoepp and Swank (1997) found that long-term sampling of forested sites showed temporal variation and patterns were not always consistent in direction of change. Comparing sites that underwent change in vegetation community cover with sites that remained

forested, Mueller and Kogel-Knabner (2009) found that forested sites that remained in forest had greater stocks of total C. They attributed this to the high content of C that is physically associated with soil clay particles in the forested sites, C that was therefore protected from further decomposition. Differences in total soil C have all been attributed to differences among vegetation types, site management, and site history. Paul et al. (2003) linked a C accounting model with forest growth models and a soil C model to examine the factors controlling total C change over time in forest plantations. They concluded that overall differences in total mass and quality of organic matter input to soils regulated rates of C accumulation. This supports the hypothesis that initial changes in total C following harvest are the result of slash inputs, changes in rooting density, and rates of decomposition (Fahey et al. 1988). Mattson and Swank (1989) found that 6 years after cutting, fine-root biomass on WS 7 was only 65% of that of WS 2, the adjacent reference watershed. This decrease could be the result of extensive root mortality and/or changes that occurred in vegetation composition and age after stand harvest. Changes in root density following harvest were evident in a study by Edwards and Ross-Todd (1983). They found decreased in situ soil CO₂ efflux rates in forested sites after harvest and concluded it was the result of reduced root respiration. In laboratory studies they found that respiration from soil alone was greater in soils from harvested sites (Edwards and Ross-Todd 1983). Scott et al. (2004) looked at changes in ecosystem C after harvest using eddy covariance measurements, examining a site preharvest and after removal of 30% above-ground biomass. This cutting intensity resulted in a 40% decline in litterfall and LAI, resulting in net ecosystem exchange of CO₂ decline of 18% the first year. Their data showed a slight decline in soil respiration.

Soil Nitrogen Availability and Mobility

Forest harvesting or disturbance often results in increased soil N availability and increased rates of N transformation processes, mineralization and nitrification (Waide et al. 1988; Donaldson and Henderson 1990; Smethurst and Nambiar 1990; O'Connell et al. 2004; Lapointe et al. 2005). Some of these studies have found increased response with increasing level of disturbance response. Lapointe et al. (2005) examined a chronosequence, looking at trembling aspen to late successional boreal forests and found that partial harvest resulted in less increase in N mineralization potential compared to clearcutting. In a study examining the relationships of site quality and N responses, O'Connell et al. (2004) looked at soil N responses to logging residue in short rotation Eucalyptus plantations. They found that inorganic N concentrations increased for 2 years following harvest, until new vegetation developed. At the more fertile sites, inorganic N was present mostly as NO₃ and in areas where logging residue was left on site, NO₃ concentrations were lower. The impact of residue was less pronounced at the low N site.

Our data show a continued increase in N availability in surface soils of WS 7 more than 20 years following harvest (table 4.3). We attribute this not only to the large input of logging residue as described above but also the changes in

vegetation community structure that have occurred since harvest. Elliott et al. (1997) and Elliott et al. (see chapter 2, this volume) described the change in forest composition through succession following clearcutting WS 7. *Quercus* and *Carya* spp. dominated the preharvest forest, while the postharvest forest is dominated by *Liriodendron*, *Robinia*, and *Acer*. Changing vegetation following harvest can result in changes in nutrient demand (Hendrickson 1988) and water demand (see Swank et al., chapter 3, this volume). Vegetation composition also directly impacts soil nutrient content and nutrient transformation processes. Knoepp et al. (2000) examined soil chemical and biological characteristics across the vegetation and elevation gradient within the Coweeta basin. They found that vegetation was the dominant regulating factor of soil biological indices, such as soil respiration and N transformations. In the northeastern United States, Mitchell (2011) examined the relationship between site N export and other site factors. He found that while N deposition played a role, soils and vegetation were also an important factor. Soil Ca availability was a determining factor in vegetation composition and the abundance of sugar maple was related to high rates of N mineralization and nitrification.

In this study, we found increased soil solution $\text{NO}_3\text{-N}$ and Ca concentrations 7 years postharvest in WS 7 30-cm tension lysimeters compared to WS 2 (figure 4.12). Ca concentrations in soil solutions collected at 100 cm were also significantly greater in WS 7 compared to WS 2 (figure 4.13). These data, along with stream chemistry data (see Swank et al., chapter 3, this volume), suggest that nutrient mobility increased following harvest. Nutrient movement is related to both the disturbance response and changes in water movement through the soil profile. As previously discussed, following the site harvest, we measured increased soil N availability as noted by greater $\text{NO}_3\text{-N}$ and $\text{NH}_4\text{-N}$ concentrations as well as potential N transformation rates, plus increased soil cation concentrations. Knoepp and Clinton (2009) examined changes in soil N availability and NO_3 in soil solution following site harvesting. They found increases in N only in harvested areas, and stream $\text{NO}_3\text{-N}$ concentrations increases only in sites where all riparian zone vegetation had been removed.

The relationship between nutrient movement and water movement through forest ecosystems has been studied through the impacts of throughfall quantity and chemistry on soil solution and stream chemistry. Armbruster et al. (2002) examined Mg deposition patterns in throughfall and outputs in soil solution and streamflow in 71 forest ecosystems across Europe. They found a gradient of deposition, with the highest Mg inputs near the ocean but increased input up to 200 km inland. Soil solution and stream output fluxes were greater where inputs in throughfall were greatest with minor influence of site soils and bedrock. Several researches experimentally decreased the throughfall inputs to the forest floor (Johnson et al. 2002a; Neiryck et al. 2002) finding concomitant declines in soil water and nutrient fluxes. Johnson et al. (2002a) also experimentally increased throughfall amounts and found that while fluxes were greater than the artificially dry site, there was no significant difference from the ambient site.

Summary

We examined the effects of forest harvesting on nutrient inputs as slash and throughfall as well as responses of total soil C and N, extractable soil cations, nitrogen availability, and movement of nutrients out of the soil through leaching. Changes of nutrient inputs, due to the input of logging residue, and changes in throughfall resulted in increased total C and N and extractable cation pools in surface soils. These increases were not long-lived, although slash residue remained on site. On the other hand, soil extractable N and potential rates of N mineralization and nitrification remained elevated for more than 20 years after harvest. We concluded that changes in vegetation following harvesting have resulted in increased rates of N cycling. This continued rate of N cycling is evident not only in the soil N processes but also in continued stream export of $\text{NO}_3\text{-N}$ (see Swank et al., chapter 3, this volume).

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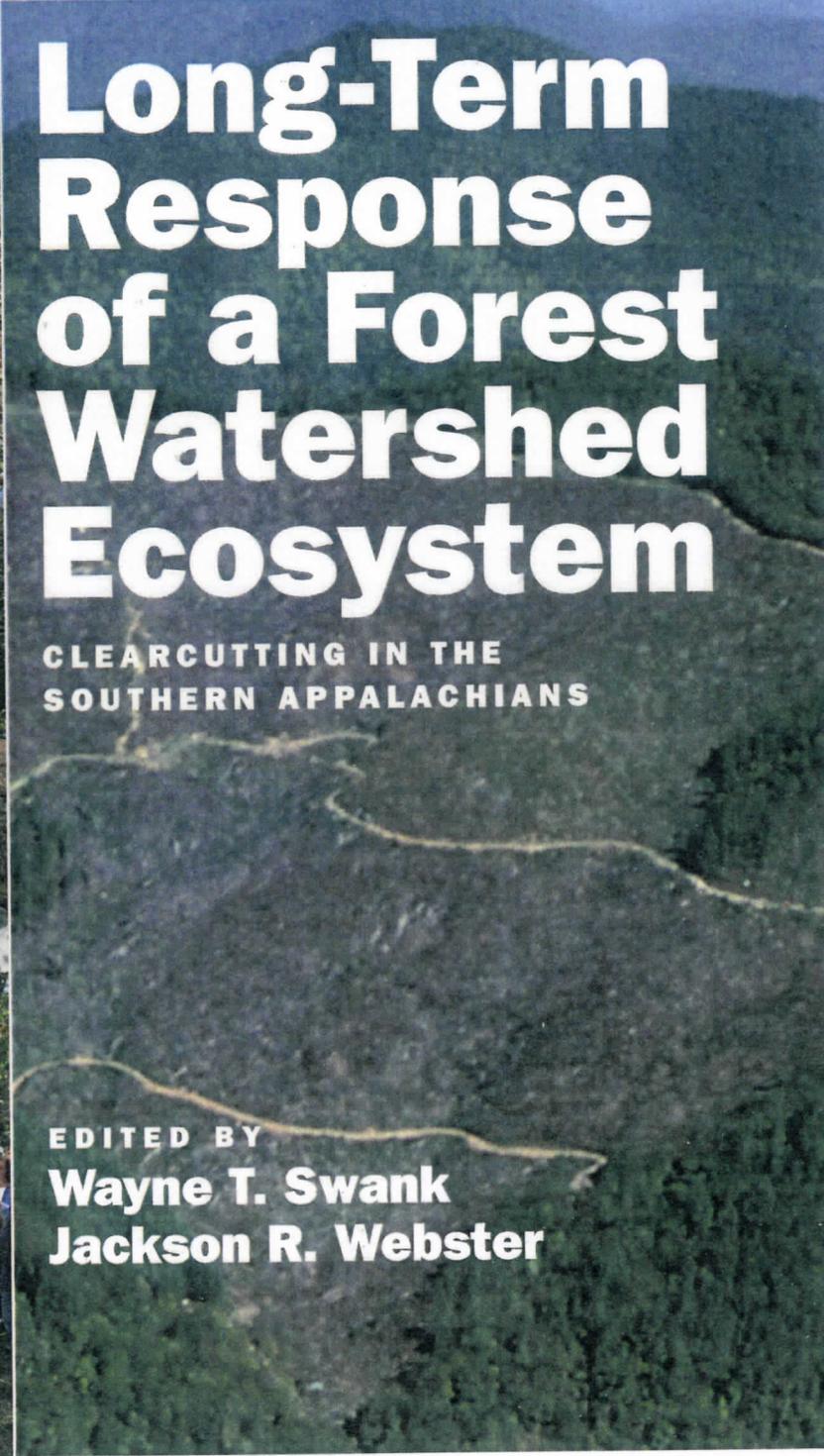
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Long-Term Response of a Forest Watershed Ecosystem

CLEARCUTTING IN THE
SOUTHERN APPALACHIANS

EDITED BY
Wayne T. Swank
Jackson R. Webster

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"No serious student of forest hydrology or ecology can survive long without encountering the name "Coweeta." The Coweeta Hydrologic Laboratory in North Carolina has rightly become world-famous across a broad spectrum of environmental science. It is well over 20 years since the last compilation of Coweeta research appeared in book form, and this volume provides a very welcome update."

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"Forest watershed research is reaching an age when some long-term trends—or the lack of them—can be evaluated. Aside from its great value as a synthesis of a comprehensive long-term research project in and of itself, this volume is a welcome scientifically objective investigation of the long-term effects of forest harvesting. This volume should reside on the bookshelves of scientists (both basic and applied), educators, policy makers, and environmental advocates.

—Dale Johnson, Emeritus Professor, University of Nevada

"This volume is a most compelling case on the value and necessity of long-term research on ecological patterns and processes. Findings summarized here are applicable way beyond the ecology and management of southern Appalachian hardwoods, by providing a framework on improving both economic and ecological values with appropriate forest management practices."

—Donald J. Leopold, Chair, Department of Environmental and Forest Biology, SUNY-ESF

Our North American forests are no longer the wild areas of past centuries; they are an economic and ecological resource undergoing changes from both natural and management disturbances. A watershed-scale and long-term perspective of forest ecosystem responses is requisite to understanding and predicting cause and effect relationships. This book synthesizes interdisciplinary studies conducted over thirty years, to evaluate responses of a clear-cut, cable-logged watershed at the Coweeta Hydrologic Laboratory in the Nantahala Mountain Range of western North Carolina. This research was the result of collaboration among Forest Service and university researchers on the most studied watershed in the Lab's 78-year history. During the experiment, a variety of natural disturbances occurred: two record floods, two record droughts, a major hurricane, a blizzard of the century, major forest diseases, and insect infestations. These disturbances provided a unique opportunity to study how they altered the recovery of the forest ecosystem. This book also shows that some long-term forest trends cannot be forecast from short-term findings, which could lead to incorrect conclusions of cause and effect relationships and natural resource management decisions.

Wayne T. Swank is Scientist Emeritus, Coweeta Hydrologic Laboratory, Southern Research Station, USDA Forest Service.

Jackson R. Webster is Professor of Ecology in the Department of Biological Sciences at Virginia Polytechnic Institute and State University.

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