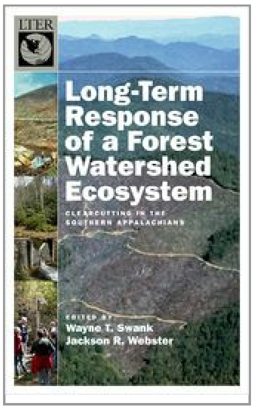


## Long-Term Response of a Forest Watershed Ecosystem: Clearcutting in the Southern Appalachians

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# Response and Recovery of Water Yield and Timing, Stream Sediment, Abiotic Parameters, and Stream Chemistry Following Logging

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## [–] Abstract and Keywords

In 1977, Watershed 7 (WS 7) at the Coweeta Hydrologic Laboratory was clearcut and logged using a mobile cable system that could access logs up to 300 m from a road and suspend the logs completely above the ground for transport to the logging deck. Watershed (WS) 2, a 12.6-ha watershed adjacent to WS 7, served as the experimental control. This chapter (1) summarizes and evaluates the long-term hydrologic and water quality responses to forest management; and (2) links stream responses with process

level research conducted within the watershed.

*Keywords:* watershed ecosystem analysis, water quality, forest management, stream response, resource sustainability

## Introduction

Watershed ecosystem analysis provides a scientific approach to quantifying and integrating resource responses to management (Hornbeck and Swank 1992) and also to address issues of resource sustainability (Christensen et al. 1996). The philosophical components of the research approach at Coweeta are (1) the quantity, timing, and quality of streamflow provides an integrated measure of ecosystem response to land management practices and (2) response to disturbance provides a valuable tool for interpreting ecosystem behavior (Swank and Crossley 1988).

Our objectives in this chapter are to (1) summarize and evaluate the long-term hydrologic and water quality responses to forest management and (2) link stream responses with process level research conducted within the watershed.

The details of general and specific forest study sites, experimental design, management prescriptions, and natural disturbances spanning the 32-year history of the study at Coweeta are described by Swank and Webster in chapter 1 of this volume. Briefly, a 59-ha south-facing mixed hardwood covered watershed was clearcut and logged in 1977 using a mobile cable system that could access logs up to 300 m from a road and suspend the logs completely above the ground for transport to the logging deck. Watershed (WS) 2, a 12.6-ha watershed adjacent to WS 7 served as the experimental control for assessing hydrologic and water quality responses to the treatment on WS 7.

## **(p.37)** Hydrology

### Methods

#### **Precipitation Input**

Precipitation inputs were measured using standard rain gages located within the Coweeta basin. Precipitation input for each watershed was calculated using established relationships between specific watershed locations within the basin and individual or multiple rain gages.

#### **Streamflow and Annual Water Yield**

We used the paired or control catchment method of analysis (Hewlett et al. 1969) to quantify the effects of logging treatment on the quantity, timing, and quality of streamflow. In this method, the relationship of stream attributes between reference and treated watersheds for the calibration period is determined by regression analysis which incorporates experimental control for climatic and biological variations within and between years. The calibration period for hydrologic analysis in this study spanned 11 years, from 1966 to 1976, with continuous measurement of discharge using sharp-crested V-notch weirs (figure 3.1). Mean annual discharge from WS 7 during this period averaged 106 cm and ranged from 76 to 149 cm.


*Figure 3.1* Upstream view of 90º V-notch weir installation on WS 7, November 2012.
(USDA Forest Service photo)

**(p.38)** The coefficient of determination ($r^2$) for total annual flow between WS 7 and WS 2 during the calibration period was 0.99. The error term ($p < 0.05$) for predicted individual annual flows for treatment years averaged ±5 cm. Regression analysis using monthly flow data was used to quantify the within-year changes; $r^2$ during the calibration period ranged from 0.96 to 0.99.

#### Early Postharvest Slash Interception Loss

An important component of evapotranspiration in forests is interception loss, which has seldom been studied after logging. Forest canopies intercept and alter the amount and chemistry of precipitation as water passes through foliage (throughfall) or flows down the stem (stemflow). Clearcutting and harvest on WS 7 removed the canopy structure for several years and added a large quantity of woody residue to the forest floor. Several (1 yr and 8 yr after cutting) studies were conducted to quantify the effects on interception loss (this chapter) and on nutrient leaching (see Knoepp et al., chapter 4, this volume).

The basic experimental design of the first study was the establishment of eighteen 4 x 4 m plots at 9 locations in WS 7, which were stratified to proportionally represent the forest types before clearcutting. One 2 x 2 m plot was nested in one corner of each 4 x 4 m plot; thus, there were a total of 36 plots. In the first year after cutting, coarse wood (CW; i.e., logs and branches with diameters ≥ 5 cm) was measured for end diameters and length of CW lying within each 4 x 4 m plot to calculate volume and surface area. Disks were taken from representative logs to determine wood density and mass. CW sampling was repeated in years 6, 7, and 11 in a long-term study of wood decomposition on WS 7 (Mattson et al. 1987; see also Mattson and Swank, chapter 7, this volume).

Fine wood (branches and stems < 5 cm in diameter) was sampled in the 2 x 2 m plots nested in the corner of each CW plot. These plots were selected to represent the range of slash dominated by stems, “brush,” or mixed slash, all < 5 cm in diameter. Wood was also sampled on these plots to estimate surface area and biomass of fine wood.

Throughfall was collected by inserting 15 x 200 cm V-shaped aluminum troughs beneath the slash, attached to 19-L polypropylene collection jugs (see figure 4.1, in chapter 4, this volume). Samples were collected on a storm event basis and volumes used to estimate interception loss and leaching of nutrients from slash.

#### Later Postharvest Forest Interception Loss

A detailed study of interception loss, atmospheric deposition, and foliage leaching was conducted on WS 7 when the regenerating forest was 8 years old (Potter et al. 1991; Potter 1992) (figure 3.2). Interception findings are reported here and canopy nutrient fluxes are reported by Knoepp et al. (see chapter 4, this volume). Three 20 x 20 m plots were located near the middle of WS 7 in a chestnut oak (*Quercus prinus*) community. A 10-m tower was located in the middle of the study site and instrumented to collect incident rainfall and dry particulate inputs to the canopy. **(p.39)**

*Figure 3.2* WS 7 eight years after clearcutting. (USDA Forest Service photo)

Thirty troughs (1.0 x 0.1 m) were randomly placed in the three plots to collect throughfall; stemflow was measured in nine 1 x 2 m plots (Potter et al. 1991). Data were collected on a storm-event basis and included 20 storms, 14 during the growing season and 6 during the dormant season, throughout the period July 1984 through August 1986.

### Results and Discussion

#### Initial Water Yield and Interception Responses

In 1978, the first full water year (May to April) following logging, streamflow increased 26.5 cm, or about 28% above the flow expected if the forest had not been cut (figure 3.3). In subsequent years, annual discharge increases declined at a rate of 5 to 7 cm per year until the fifth year after cutting, when annual flow was just 4 cm above pretreatment levels. Thereafter, changes in flow were not significant ($p > 0.05$) and discharge fluctuated around expected baseline values.

The pattern of initial response and early recovery of annual streamflow after clearcutting WS 7 are consistent with other forest cutting experiments at Coweeta (Swank et al. 1988) and in other locations of the Appalachian region of the United States (see Adams and Kochendenfer, chapter 12 and Hornbeck et al., chapter 13, this volume). Water yield increases are typically greatest in the first year after cutting because transpiration is most reduced due to minimal leaf area index (LAI). In subsequent years, as sprouts and seedlings regrow, LAI and transpiration increase (see Boring et al., chapter 2, this volume) resulting in a logarithmic decline in streamflow over the first six years of succession.

**(p.40)**

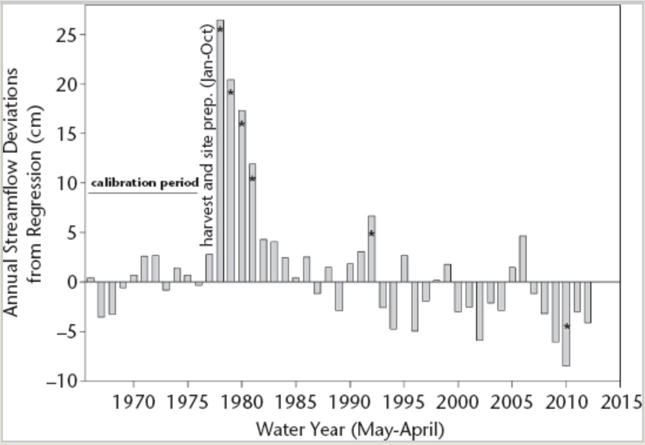


*Figure 3.3* Annual deviations in streamflow on WS 7 prior to and following clearcutting and commercial logging at Coweeta Hydrologic Laboratory, 1967–2012. * Denotes significant change; $p < 0.05$).

Logging slash interception was measured for a total of 36 storms, ranging from 5 to 12 mm from December 1977 through April 1979. Interception loss (precipitation minus throughfall) was a linear function of rainfall amount for all types of slash. Statistical analyses of regression slopes showed no significant differences among the slash types < 5 cm, so all data from the small material were combined in the regressions. However, the regression slope of the coarse wood (logs and branches ≥ 5 cm) was different from all of the other types of slash and resulted in a different regression equation.

Linear regressions for estimating interception loss from slash are:

(Eq. 1) $\mathrm{IL}\ (\mathrm{CWD}) = -0.44 + 0.2098\ (\mathrm{P});\ r^2 = 0.70$

$$\mathrm{IL}\,(\mathrm{CWD}) = -0.44 + 0.2098\,(\mathrm{P})\,; r^2 = 0.70$$

(Eq. 2) $\mathrm{IL}\ (\mathrm{other\ slash}) = -0.82 + 0.1516\ (\mathrm{P});\ r^2 = 0.47$

$$\mathrm{IL}\,(\mathrm{otherslash}) = -0.82 + 0.1516\,(\mathrm{P})\,; r^2 = 0.47$$

where IL is interception loss and P is precipitation, both in mm.

Estimates of annual interception loss using Eqs. 1 and 2 were derived for the period

September 1977–August 1978; storm precipitation was measured monthly on a gage located on WS 7 and adjusted for each month based on previously developed seasonal weighting factors (Swift et al. 1988). Precipitation for the 12-month period totaled 1,731 mm, with a monthly range of 22 to 265 mm delivered in 76 storms over the year. Interception loss for CW was estimated to be 324 mm, or 18.7% of precipitation, compared to an estimated interception loss of 197 mm, or 11.4% of precipitation for smaller slash. A combined wood interception loss was derived by weighting the amount of the two slash types measured on the 36 plots based on wood surface area. The mean wood surface area per plot area was 0.448 $m^2/m^2$ for CW and 0.898 $m^2/m^2$ for smaller slash, for a total wood surface area of **(p.41)** 1.346 $m^2/m^2$. Using the appropriate weighting factors, annual interception loss for wood was estimated to be 239 mm, or 13.8% of precipitation.

Similar studies of interception by logging residue are not available for comparison with these findings. However, Helvey and Patric (1988) reported an annual interception loss for mature hardwoods at Coweeta of 250 mm, or 13% total precipitation, including 3% litter interception loss. Thus, the early postharvesting interception component of the hydrologic budget on WS 7 was not substantially altered and the first several years of annual water yield increases measured after cutting and harvest (figure 3.3) were mainly due to reductions in transpiration. There are several factors contributing to the initial high interception loss. First, only sawlogs were harvested and the remaining vegetation was cut as part of the site preparation treatment, which contributed to a high loading of woody residue. Secondly, WS 7 is a south-facing slope; previous research (Swift 1972) found radiation available for evapotranspiration is greater on south- than north-facing slopes, an important factor that was included in a model used to predict water yield response on WS 7 (table 3.1).

#### Later Postharvest Forest Interception Loss

In the regenerating 8-year-old forest (Potter et al. 1991; Potter 1992), data were combined for all storms. Throughfall and stemflow were estimated to be 83.3% and 5.8% of precipitation, respectively, for an estimated interception loss of 10.9% of precipitation. These water fluxes for the regenerating stand were similar in magnitude to those that were concurrently measured in the adjacent reference hardwood forest on WS 2 (Swank et al. 1992).

The recovery of canopy interception loss early in succession can be mainly attributed to a rapid recovery of leaf area on WS 7 (see Boring et al., chapter 2, this volume). The contributions of interception loss to watershed evaporotranspiration (Et) and streamflow recovery is a dynamic process. The importance of interception loss from wood measured the first year after cutting declined over time with decomposition. However, 11 years after cutting, the quantity of CW was still significant (see Mattson and Swank, chapter 7, this volume), which contributed to

**Table 3.1 Comparison of annual observed vs. predicted increase in water yield following clearcutting on Coweeta WS 7.**

| Year following clearcutting | Increased predicted by model (cm) | Observed Increase (cm) |
|---|---|---|
| 1 | 25 | 26 |
| 2 | 17 | 20 |
| 3 | 12 | 17 |
| 4 | 8 | 12 |
| 5 | 6 | 4 |
| 6 | 3 | 4 |
| Total | 71 | 83 |

*Source*: Swank et al. (1988).

**(p.42)** water storage and interception loss from the forest floor. Over the same time period, canopy interception loss was returning to precutting levels. Thus, during the first decade of regrowth, the combined interception loss from wood and the regrowing canopy probably exceeded that for a mature hardwood forest and may have contributed to some of the variability in annual water yield (figure 3.3).

#### Longer-Term Water Yield Responses

The significant increase in streamflow in 1992, when the forest was 15 years old (figure 3.3) has been attributed to a reduction in both stem density and LAI associated with competition and self-thinning (stem exclusion stage) of rapid growing coppice vegetation (Swank et al 2001; Elliott et al. 1997), a high mortality rate of *Robinia pseudoacacia* caused by stem borers (see Boring et al, chapter 2, this volume), decline of dogwood (*Cornus florida*) due to a disease, dogwood anthracnose (Chellemi et al. 1992), and loss of abundant American chestnut (*Castanea dentata*) sprouts due to the chestnut blight (see Boring et al., chapter 2, this volume). Similar patterns of changes in stand structure, water use, and streamflow have been found in other clearcutting experiments at Coweeta (Swift and Swank 1981).

Canopy openings created during the stem exclusion stage of succession were short-lived; and by 1994 to 1995, 17 years after cutting, the stand basal area was 23 $m^2$/ha, which is similar to the 25 $m^2$/ha basal area of the original forest (Elliott et al. 1997) and LAI also increased to precutting levels (see Boring et al., chapter 2, this volume). In the ensuing decade after LAI stabilized, there has been a pattern of annual streamflow reductions that frequently exceed 2.5 cm (figure 3.3). We hypothesize that higher Et for the regrowing versus mature forest is related to higher transpiration loss associated with major shifts in species composition. For example, there were very large increases in basal area of *Liriodendron tulipifera*, *Acer rubrum*, and *Robinia pseudoacacia* in the successional forest and an equally large decline for combined *Carya* and *Quercus* spp. (see Boring et al., chapter 2, this volume). This hypothesis is supported by recent physiological research at Coweeta that has shown large differences in canopy transpiration rates among hardwood species. Specifically, diffuse porous species, such as yellow popular and red maple, have much higher transpiration rates compared to oak and hickory species (Ford et al. 2010; Ford et al. 2011). Persistent decreases in annual water yield were also observed at Hubbard Brook following a harvest and attributed to higher transpiration rates for ring porous early succession species compared to mature northern hardwood forests (Hornbeck et al. 1977; see also Hornbeck et al., chapter 13, this volume).

### Water Yield Model

One of the original objectives of the cutting experiment on WS 7 was to obtain water yield response data for a south-facing watershed. Previous syntheses of watershed experiments in the Appalachian Highlands Physiographic region (Douglass and Swank 1972, 1975) established empirical equations between first-year water yield **(p.43)** increases as a function of percent basal area cut and an insolation index (energy variable related to slope aspect) for a catchment. In addition, another empirical equation was developed (Douglass and Swank 1975) for predicting water yield increases for any year following harvest until streamflow returns to baseline levels. The models contain little data from south-facing clearcut watersheds with natural forest succession. However, predictions from these models generally showed good agreement with annual water yield responses measured on south-facing WS 7 (table 3.1). The first year after cutting, streamflow increased about 26 cm compared to 25 cm predicted by the model. Subsequently, in the next 2 years, predicted increases were substantially below observed increases. These years coincided with the wettest year on record and one of the driest years at Coweeta. In the ensuing 3 years, predictions were in close agreement with observed increases in water yield and the total change predicted for the 6-year period was within 17% of the observed change.

### Intra-Annual Water Yield

During the first 3 postcutting years, the monthly distributions of water yield increases (figure 3.4) were similar to other low-elevation cutting experiments at Coweeta (Swank et al. 1988). Flow increases occurred in every month, with the smallest amounts in the spring (April and May), at the same time that soil moisture in an undisturbed forest is usually fully recharged. Concurrent with the growing season, streamflow increases become larger due to reduced Et on WS 7. Substantial flow increases continued into the late fall and winter months and partially reflect the lag between when reduced Et occurs on the cut watershed and when the water savings reach the weir during the period of high precipitation (figure 3.4).

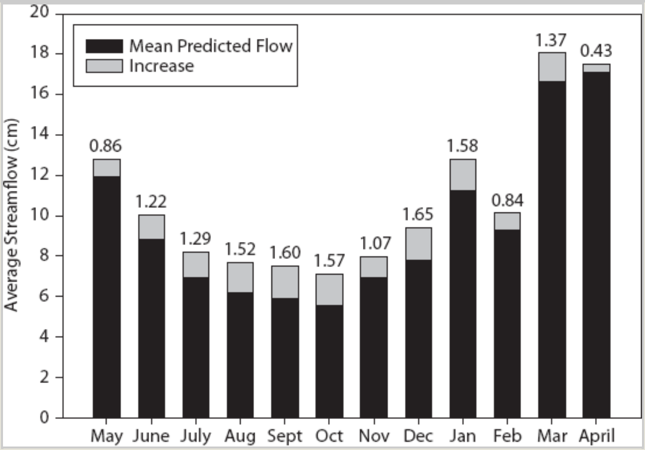


*Figure 3.4* Mean monthly changes in water yield during the first 4 postcutting years on Coweeta WS 7.

(Modified from Swank et al., 1982)

**(p.44)** Of notable importance is that two of the largest flow increases (1.6 cm, or 28% increase) occurred in September and October when flows are normally low and water demands are high.

### Storm Hydrograph Responses

Detailed analysis of eight storm hydrograph parameters was conducted for WS 7 using pretreatment storm data on WS 7 regressed against the same parameters as the reference catchment (WS 2). The analysis used data for 75 storms (≥ 2 cm) from the first 4 years after treatment, which encompassed the period of maximum water yield increase (Swank et al. 2001).

Following harvest, statistically significant changes in regression intercepts and slopes were found for all storm parameters except time to storm peak (table 3.2). The largest increases occurred in peak flow rates (15%) and initial flow rates (14%); the latter is due to elevated rates of baseflow from the watershed. Quickflow (stormflow) volume increased 10%, which was associated with a 10% increase in recession time. Taken collectively, the hydrograph responses are considered to be of minor importance to downstream flooding. For example, in the first 4 years after cutting and harvest, the average precipitation storm, ≥ 2 cm, increased the quickflow volume by only 0.03 cm, or 2.43 $m^3$/ha, and the peakflow rate by 1.7 $m^3$/ha.

Storm hydrograph responses to harvest are partly dependent upon (1) the magnitude and method of logging and associated road disturbance and (2) the inherent responsiveness of the watershed to precipitation events in the absence of

**Table 3.2 Storm hydrograph parameters and changes during the first 4 years following clearcutting and logging on WS 7.**

| Parameter | Mean for treatment watershed | Significance of regression coefficients | Percent change in parameter after treatment for mean storm |
|---|---|---|---|

| | WS 7 | WS 2 | Intercept | Slope | |
|---|---|---|---|---|---|
| Initial flow rate ($m^3$ $s^{-1}$ $km^{-2}$) | 0.037 | 0.27 | ** | ** | 14 |
| Peakflow rate($m^3$ $s^{-1}$ $km^{-2}$) | 0.136 | 0.127 | ** | ** | 15 |
| Time to peak (h) | 8.0 | 8.0 | NS[b] | NS | 0 |
| Total quickflow volume (cm) | 0.32 | 0.40 | ** | ** | 10 |
| Quickflow after peak (cm) | 0.08 | 0.09 | ** | ** | 6 |
| Quickflow after peak (cm) | 0.24 | 0.31 | * | NS | 11 |
| Quickflow duration (h) | 27.6 | 28.3 | ** | * | 5 |
| Recession time (h) | 20.0 | 21.0 | ** | NS | 10 |

[a] Derived from difference between value predicted from calibration regression and measured value.

([b]) Nonsignificant.

(*) $p < 0.05$.

(**) $p < 0.01$.

*Source:* Modified from Swank et al. (1982).

**(p.45)** disturbance. Inherent responsiveness is driven by a variety of physical factors such as watershed size, soil depth, slope and topographic complexity, and infiltration rates. The response factor (mean annual quickflow/mean annual precipitation) for WS 7 was very low (0.04), which accounts for some of the small changes in storm hydrograph parameters. Furthermore, the low density of logging roads, minimal disturbance of the surface soil by cable logging (see Swank and Webster, chapter 1, this volume), and careful design of roads (Swift 1988) also limited changes in stormflow on WS 7.

## Abiotic Responses to Clearcutting

### Soil Moisture and Temperature

Regeneration cutting can produce significant changes in the microenvironment of the forest floor and soil that in turn regulate ecosystem processes, such as decomposition; microbial activity; nutrient cycles; and the germination, sprouting, survival, and growth of vegetation. Beginning in August 1977, studies were initiated on WS 7 to evaluate the effects of harvesting on soil moisture and temperature. Soil moisture was measured at biweekly intervals in the $O_1$ and $O_2$ (organic) litter layers of the forest floor and 0 to 10-cm and 10–30-cm depths in the soil on WS 7 and also WS 2, the adjacent control watershed (Swank and Vose 1988). In the first autumn (Aug–Nov) after harvest, litter moisture was 20% to 30% below that found on WS 2 (table 3.3). In the subsequent winter quarter, litter moisture was similar on both watersheds. However, in the ensuing year (1978), litter moisture was consistently 30% to 50 % lower in the clearcut compared to WS 2. In contrast, surface soil moisture (0–10 cm); increased more than 11% the first year after cutting (table 3.3). Deeper in the soil profile, moisture increases were small or showed no changes.

Soil temperatures were measured at the litter-soil interface on WS 7 during the precut year (1976) prior to clearcutting and in the first growing season after cutting and logging (Swank and Vose 1988). Mean monthly temperatures were 7°C to 10°C above precut levels in the period May through October in the first year after cutting. Mean monthly maximum soil temperatures showed large increases with values being 10°C to 35°C above precut levels and daily maximum temperatures frequently exceeded 45°C during this period. In subsequent years, increases in surface soil temperatures were moderated by shade from regeneration.

## Stream Chemistry

### Methods

Stream chemistry measurements began in late 1971 on both WS 7 and WS 2. Weekly grab samples have been collected at a fixed location just above the weir from each watershed since 1971, and flow proportional samples were also collected in the period 1975–1981. Solute determinations include $NO_3^-$, $NH_4^+$, $SO_4^{-2}$, $PO_4^{-3}$,$Cl^-$, and **(p.46)**

**Table 3.3 Quarterly forest floor and soil moisture averages (percent) for mixed hardwood forests during the first year after clearcutting (WS 7) and for the reference watershed (WS 2).**

| Treatment and depth | Year and Quater | | | |
|---|---|---|---|---|
| | Aug–Nov 1977 | Dec–Mar 1977–78 | Apr–Jul 1978 | Aug–Nov 1978 |
| | Water content (percent by weight) | | | |
| Clearcut | | | | |
| $O_1$ | 70 | 124 | 88 | 51 |
| $O_2$ | 95 | 170 | 97 | 58 |
| 0 to < 10 cm | 46 | 70 | 55 | 31 |
| 10 to 30 cm | 36 | 40 | 34 | 23 |
| Control | | | | |
| $O_1$ | 92 | 121 | 120 | 105 |
| $O_2$ | 126 | 211 | 120 | 109 |

| | | | | |
|---|---|---|---|---|
| 0 to < 10 cm | 35 | 59 | 36 | 28 |
| 10 to 30 cm | 32 | 40 | 29 | 24 |

*Note*: $O_1$ = includes fresh or slightly decomposed organic materials.

$O_2$ = includes intermediate and highly decomposed organic materials.

*Source*: Modified from Swank and Vose (1988).

base cations ($Ca^{+2}$, $Mg^{+2}$, $K^{+}$, $Na^{+}$) using established analytical methods at Coweeta (Brown et al. 2009). Comparison of 3-year average annual export of solutes calculated from weekly grab samples versus flow-proportional samples showed good agreement for most solutes (Swank and Waide 1988). Annual change in export of each solute due to treatment the first 6 years was estimated from pretreatment regressions of monthly exports between WS 7 and WS 2. Relationships of monthly exports between the two catchments were good ($r^2$ values ≥ 0.92) for most ions.

### Response to Treatment

Stream chemistry responses to treatment were relatively small (table 3.4) as described in an earlier analysis (Swank et al. 2001). Increases in export of $PO_4$, K, Ca, and Mg in the first full year following logging are partially related to release from the fertilizer applied to roads. Lack of significant $NO_3$ response was due in part to sediment denitrification that depleted $NO_3$ before it reached the weir (Swank and Caskey 1982). Denitrification in Big Hurricane Branch (WS 7) was remeasured in 2004 as part of a large regional study in 49 streams with varying land-use categories (Mulholland et al. 2009). They found measureable but lower rates of denitrification than those found in 1977 by Swank and Caskey (1982). However, differences in both methods and the supply of $NO_3$ of the two studies limit direct quantitative comparisons.

The magnitude of nutrient export is determined by both changes in solute concentrations and increases in discharge resulting from reduced Et following cutting. The first two years after cutting, annual flow increased an average of 23.5 cm/y, but **(p.47)**

**Table 3.4 Annual changes in streamflow and solutes following clearcutting and logging (posttreatment–pretreatment) on WS 7.**

| Time since treatment (May–April water year) | Flow (cm) | Increase or decrease in streamflow and solute export (kg $ha^{-1}$)[a] | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|
| | | $NO_3$-N | $NH_4$-N | $PO_4$-P | K | Na | Ca | Mg | $SO_4$-S | Cl |
| First 4 months | 0.5 | 0.01 | < 0.01 | 0.01 | 0.43 | 0.42 | 0.24 | 0.26 | 0.39 | 0.68 |
| First full year | 26.5 | 0.26 | 0.03 | 0.04 | 1.98 | 1.37 | 2.60 | 0.96 | 0.27 | 1.13 |
| Year 2 | 20.5 | 1.12 | < 0.01 | 0.01 | 1.95 | 2.22 | 2.51 | 1.15 | −0.08 | 1.62 |

| | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|
| Year 3 | 17.3 | 1.27 | 0.05 | 0.02 | 2.40 | 2.68 | 3.16 | 1.42 | 0.39 | 2.08 |
| Year 4 | 11.9 | 0.25 | 0.15 | 0.02 | 0.80 | 1.07 | 1.63 | 0.46 | 0.31 | 0.59 |
| Year 5 | 4.3 | 0.28 | 0.01 | < 0.01 | 0.52 | 0.13 | 1.19 | 0.18 | 0.04 | 0.10 |
| Year 6 | 4.1 | 0.62 | 0.06 | < 0.01 | 0.73 | 0.69 | 0.89 | 0.42 | −0.06 | 0.33 |

([a]) Annual increase or decrease derived from sum of deviations using monthly calibration regressions.

*Source*: Modified from Swank et al. (2001).

maximum concentrations of most solutes did not occur until the third year when water yield on WS 7 was still more than 17 cm above pretreatment levels (table 3.1). By the sixth year of postdisturbance, streamflow was near pretreatment levels and solute exports also appeared to be approaching pretreatment levels. The longer-term responses of most solutes showed a similar pattern. For example, small (5 µeq/L) increases in Ca concentrations were observed after logging but later returned to expected pretreatment levels (figure 3.5). Similarly, following the initial increase in K concentrations, interannual concentrations of K after 1983 were highly variable, and there were no consistent differences in concentrations between WS 7 and WS 2 (figure 3.5). The same pattern was true for Mg (figure 3.6). There was little change in $SO_4$ concentrations on WS 7 after cutting, but beginning in 1989 there has been a consistent decline in $SO_4$ concentrations on both WS 7 and reference WS 2 (figure 3.6). Concentrations of $NH_4$ and $PO_4$ were low and almost identical for WS 7 and WS 2 during the entire period of record. In contrast, the long-term record for $NO_3$ showed interesting, significant dynamics following cutting and forest succession (figure 3.7).

The initial increase in $NO_3$ concentrations on WS 7 was attributed to increases in soil N pools and concentrations the first three years after logging (Waide et al. 1988; see also Knoepp et al., chapter 4, this volume). Decline in stream $NO_3$ concentrations until about 1987 are associated with the rapid sequestration and storage of nutrients in successional vegetation (Boring et al. 1988; see also Boring et al., chapter 2, this volume). However, major shifts in internal ecosystem N cycling are evident in a large, sustained pulse of $NO_3$ to the stream from about 1987 through 1997 (figure 3.7). Mean annual peak $NO_3$ concentrations 20 years after disturbance are about double the values in the early postharvest years. Thereafter, $NO_3$ concentrations declined about 8 µeq/L over a 5-year period, followed by another increase during the ensuing 5 years that reached a maximum of 12 µeq/L in 2008 (figure 3.7).

**(p.48)**

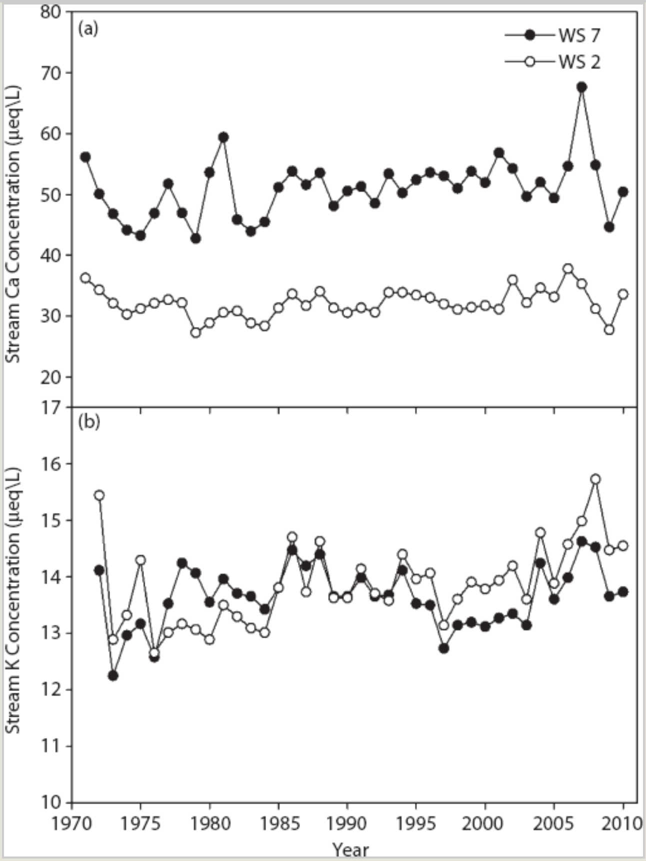


*Figure 3.5* Mean annual concentrations of Ca (a) and K (b) in streamwater of WS 7 and WS 2 during calibration (1971–1976), treatment activities (1976–1977), and postharvest period (1978–2010).

A complex combination of ecological processes contribute to the magnitude and temporal dynamics in stream $NO_3$. Accelerated $NO_3$ release to the stream coincides with stem exclusion in 1992, and thus, some of the $NO_3$ loss was probably due to the reduced uptake. However, the largest contributor to increased N availability was extensive black locust mortality due to locust borer infestation, which is a similar response observed in another early succession watershed (WS 6) at Coweeta with large locust infestations and mortality (Swank and Waide 1988). Black locust is a symbiotic nitrogen fixer; in the 4-year-old locust stands on WS 7, fixation was estimated to be 30 kg N $ha^{-1}$ $yr^{-1}$ while fixation catchment wide was estimated at 10 kg N $ha^{-1}$ $yr^{-1}$ (Boring and Swank 1984). Moreover, black locust is known to accumulate large quantities of N in foliage, roots, branches, and stems (Boring and Swank 1984). Decomposition of the N-rich organic matter from the dead trees (330 stems/ha) could be a major source of stream $NO_3$ later in succession (figure 3.7). **(p.49)**

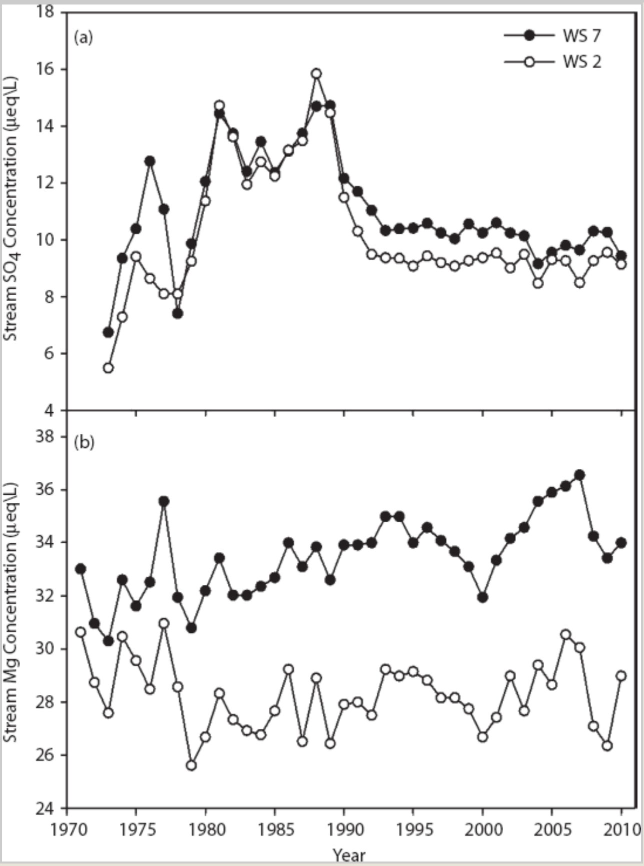


*Figure 3.6* Mean annual concentrations of $SO_4$ (a) and Mg (b) in streamwater of WS 7 and WS 2 during calibration (1971–1976), treatment activities (1976–1977), and postharvest period (1978–2010).

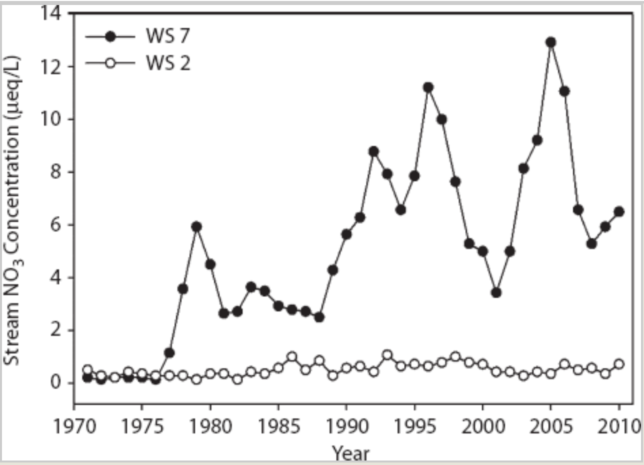


*Figure 3.7* Mean annual concentrations of $NO_3$ in streamwater of WS 7 and WS 2 during calibration (1971–1978), treatment activities (1976–1977), and postharvest period (1978–2010).

**(p.50)** Decomposing logging residue (see Mattson and Swank, chapter 7, this volume) is also a potential source of long-term stream $NO_3$ enrichment that appears in the stream. Other possible reasons for increased stream $NO_3$ concentrations are elevated rates of soil N mineralization and nitrification and reduction in the soil C/N ratio.

Measurement of the C/N ratio showed no significant change over the 18-year period following cutting (Knoepp and Swank 1997). However, long-term assessment of soil N transformations show continued increase in N availability in surface soils 20 years following harvest (see Knoepp et al., chapter 4, this volume).

The stream $NO_3$ responses observed on WS 7 demonstrate the value of long-term studies in forest ecosystems and the processes regulating system responses. For example, the first 10-year trend in concentrations suggested that $NO_3$ concentrations had returned to near pretreatment levels. However, in the subsequent 30-year period, concentrations showed very large increases and decreases that greatly exceeded the initial responses. Thus, early termination of the study and associated conclusions would have been incomplete and partly inaccurate.

The importance of the interaction between successional vegetation (e.g., black locust) and insect infestations on stream $NO_3$ is clearly evident on WS 7; however, this does not explain all of the temporal variation in stream $NO_3$. For example, interannual magnitude and variability of stream $NO_3$ concentration are also related to hydrologic variables. Annual streamflow on WS 7, ranged from 45 to 130 cm/y from 1990 to 2009, and explained 36% of the annual variation in stream $NO_3$ concentrations following cutting (figure 3.8).

Taken collectively, nutrient losses on WS 7 should not have an adverse impact on the sustainability and growth of the successional forest. Atmospheric deposition of nutrients exceeded the elevated losses of nutrients in most years of the study (Swank et al. 2001). Moreover, the high N-fixation rates of black locust and availability of N to other tree species can be viewed as a benefit to tree growth and forest health. Further discussion on the relevance of findings to management and ecological values is found in chapter 14 of this volume.

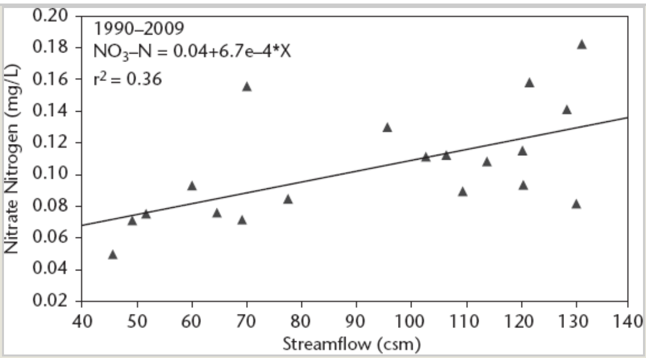


*Figure 3.8* The relationship between mean annual $NO_3$-N concentrations and annual streamflow on Coweeta WS 7 over a 20-year period.

## **(p.51)** Sediment

### Methods

The effects of the management practices on soil loss to streams on WS 7 were determined at periodic intervals by measuring sediment accumulations in the weir ponding basin and also on the approach apron of the ponding basin of both WS 7 and WS 2. This approach does not account for suspended sediment that passes across the weir blade; therefore, total sediment export was underestimated. Sediment volumes were estimated by measuring sediment elevations along permanent transects with a transit and level rod before and after cleaning the ponding basin and approach aprons. Bulk samples were collected at each elevation measurement and processed to estimate dry weight. A pretreatment calibration regression equation of periodic sediment loss over a 2-year period between WS 7 and WS 2 was derived ($r^2 = 0.91$) to estimate changes in sediment loss due to management.

Soil losses on subdrainages within WS 7 were also measured to separate and quantify sediment sources due to roads versus logging. Discharge and soil export were sampled with an H flume and a Coshocton wheel using procedures described by Douglass and Van Lear (1983). One installation was located in a perennial stream below the middle logging road (figure 3.9) and three installations were located above the influence of logging roads to evaluate effects of cutting and logging only.

*Figure 3.9* One foot H-Flume and Coshocton sampler (no. 701) on WS 7—one of four such installations used to assess water quality above and below roads, 1976.

(USDA Forest Service photo)

**(p.52)** Results

Sediment yield in the 2 years of pretreatment calibration from WS 7 and WS 2 averaged 230 and 135 kg $ha^{-1}$ $yr^{-1}$ respectively. These baseline sediment yields are similar to the mean values for small, forested catchments in the eastern United States summarized by Patric et al. (1984). In mid-May 1976, roads in WS 7 were fertilized and seeded but road fills and the running surface were unsettled and without grass or gravel cover. The third week of May 1976, a 16-cm storm occurred and was followed May 28 by a larger storm of 22 cm with intensities of 7 cm/h. The second storm produced the greatest discharge rates measured on most catchments at Coweeta during the previous 62 years of the Laboratory gaging history. These two events greatly accelerated sediment yield on both

WS 7 and WS 2 with an increase in soil loss in May on WS 7 of 1,470 kg $ha^{-1}$ $yr^{-1}$ (figure 3.10). Roads were the primary source of increased sediment yield as illustrated by soil loss measured at one of the gaging stations in a stream immediately below a road crossing the middle of the catchment (figure 3.11). Following the May storms, sediment yield at the station was nearly 50 t from 0.086 ha of road contributing area comprised of roadbed, cut, and fill. Following road stabilization and minimum use over the period of June through December 1976, soil loss from the road was low, but it accelerated again during the peak of logging activities (figure 3.10). In ensuing years, soil loss returned to baseline levels. In the same time period, samplers located above roads, which were only influenced by cutting and yarding, only collected small amounts of material comprised mainly of organic matter.

Following the initial pulse of sediment from the May 1976 storms, sediment yield showed much different temporal patterns at the weir (figure 3.10) compared to sediment loss from the roads. Sediment yield remained substantially elevated

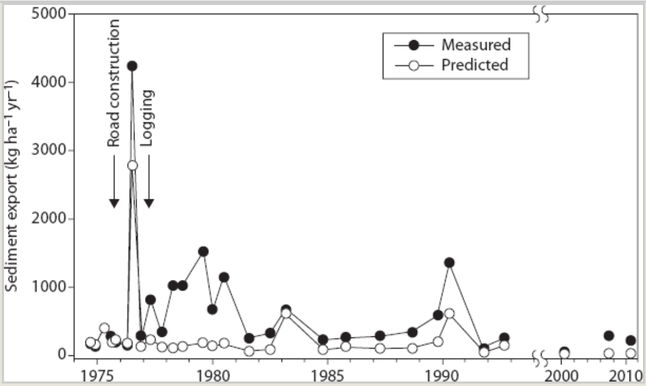


*Figure 3.10* Sediment yield measured in the ponding basin on WS 7 following logging compared to the sediment yield predicted from WS 2, the adjacent reference watershed, over a 35-year period.

**(p.53)**

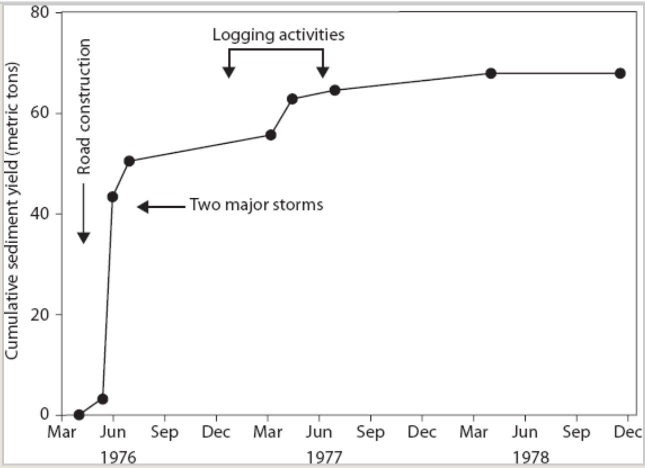


*Figure 3.11* Cumulative sediment yield measured in a first-order stream below a logging road during the first 32 months after logging.

during and after logging disturbances and continued to have sediment losses that were frequently in excess of 500 kg $ha^{-1}$ $yr^{-1}$ above expected losses over the period 1978—1985 (figure 3.10). A large pulse of sediment was delivered to weirs on both WS 2 and WS 7 in 1989 in response to the wettest year on record at Coweeta. Sediment yield on WS 7 was about 800 kg $ha^{-1}$ $yr^{-1}$ above the expected yield. In 1991 sediment yield on WS 7 returned to pretreatment levels (234 kg $ha^{-1}$ $yr^{-1}$) and remained at about the same level (179 kg $ha^{-1}$ $yr^{-1}$) based on three sample periods in subsequent years (figure 3.10).

The long-term sediment yield responses illustrate the delay or lag between pulsed sediment inputs to a stream and routing of sediments through the watershed. Soil loss derived from roads was very low following stabilization with grass and gravel cover. Moreover, following logging, road travel was minimal—roads were only used for access to research sites. Also, based on long-term cross-section measurements on the main stream on WS 7, there have only been infrequent and minor instances of stream bank erosion (Webster et al. unpublished data). Thus, in the absence of significant additional sources of sediment to WS 7 streams, the long-term sediment increases observed at the weir on WS 7 were due to a continued release of sediment from upstream storage, which was deposited from three road crossings on perennial streams and four crossings of intermittent streams on WS 7 in the May 1976 storms.

The unique conditions that produced these sediment responses should be recognized, that is, record storms occurred at the precise time when roads were freshly constructed and without vegetation cover, and thus most vulnerable to erosion. It is also important to point out that best management practices were used in harvesting **(p.54)** and in logging road location and design. The long-term effect of this management prescription on water, soil, vegetation sustainability and health, and the structure and function of benthic invertebrates is further discussed in a concluding synthesis (see Webster et al., chapter 14, this volume).

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