Biogeochemistry **38:** 303–335, 1997.

Solute dynamics in soil water and groundwater in a central Amazon catchment undergoing deforestation

MICHAEL R. WILLIAMS[1,*], THOMAS R. FISHER[1] &
JOHN M. MELACK[2]
[1]*Center for Estuarine and Environmental Studies, Horn Point Laboratories, Cambridge, MD 21613, U.S.A.;* [2]*The University of California at Santa Barbara, Department of Ecology, Evolution and Marine Biology, and Institute for Computational Earth System Science, Santa Barbara, CA 93106, U.S.A. (** *Current address: Institute for Computational Earth System Science, University of California, Santa Barbara, CA 93106, U.S.A.)*

Accepted 27 February 1997

Key words: Amazon, deforestation, hydrologic pathway, groundwater, nitrogen, rain forest, slash-and-burn agriculture, solutes, tropical

Abstract. Hydrochemical changes caused by slash-and-burn agricultural practices in a small upland catchment in the central Amazon were measured. Solute concentrations were analyzed in wet deposition, overland flow, shallow throughflow, groundwater and bank seepage in a forested plot (about 5 ha) and an adjacent plot (about 2 ha) which had been deforested in July 1989 and planted to manioc, and in stream water in partially deforested and forested catchments. Measurements were made from November 1988 to June 1990. The effects of slash-and-burn agricultural practices observed in the experimental plot included increased overland flow, erosion, and large losses of solutes from the rooted zone. Concentrations of NO_3^-, Na^+, K^+, SO_4^{2-}, Cl^- and Mn in throughflow of the experimental plot were higher than those of the control plot by more than a factor of 10. Extensive leaching occurred after cutting and burning, but solute transfers were diminished along pathway stages of throughflow to groundwater, and particularly within the riparian zone of the catchment. High concentrations of N and P in overland flow indicate the importance of using forested riparian buffers to mitigate solute inputs to receiving waters in tropical catchments.

Introduction

Estimates of deforestation from the Amazon basin using Landsat-Thematic Mapper images have indicated that about 300,000 km^2 of forest were cut from 1960 to 1989 (Tardin & da Cunha 1990). Problems associated with such large-scale deforestation include changes in regional water balance (Salati & Vose 1984), alterations of the carbon cycle (Houghton 1990), and the extinction of plant and animal species (Muller-Dombois 1990). Most deforested areas in the Amazon basin are located in western states such as Rondônia (Woodwell et al. 1986; Booth 1989), near Manaus in the central Amazon, and in eastern parts of the basin (Fearnside 1990). Accessible areas bordering navigable rivers and roads are particularly prone to deforestation (Fearnside 1989). In spite of continuing large-scale deforestation in the Amazon basin, the effects of deforestation on the ecology of adjacent streams, rivers and lakes in the Amazon have not been adequately addressed.

In the Amazon basin, slash-and-burn agricultural practices are a pervasive cause of deforestation, but little is known about the biogeochemical processes or solute fluxes that occur after this kind of disturbance. In temperate zone catchments, disturbances can reduce transpiration and increase runoff, erosion and leaching (Likens et al. 1970; Schindler et al. 1980; Swank et al. 1988), and associated effluxes of solutes, in particular N and P, can alter surface water quality (Wright 1976; Bayley & Schindler 1991; Bayley et al. 1992a, b).

Soil processes are known to be strong determinants of solution chemistry along hydrologic pathways and include adsorption and ion exchange, dissolution and precipitation reactions, and biotic uptake or release (Sposito 1989). Cation release from soils is mediated commonly by H^+, whose quantities are regulated by internal cycling and atmospheric inputs. The ionic composition of soil solutions may be controlled also by carbonic and organic acids derived from precipitation and the decomposition of organic matter (Sollins 1980). In some temperate-zone catchments, organic acids may play a dominant role in leaching of the surface horizon (Johnson et al. 1977), but they are often retained in sub-surface horizons, resulting in pH increases in systems not dominated by NO_3^- or SO_4^{2-} leaching (Ugolini et al. 1977). Both tropical- and temperate-zone studies have shown that NO_3^- production is greatest in the litter and A horizons, whereas uptake of NO_3^- may occur at depth (Gosz 1981; Matson et al. 1987; Hart & Firestone 1991). Increased concentrations of Fe and Al oxides in forest subsoils contribute to the retention of SO_4^{2-} at depth (Parfitt 1978). The high proportion of Fe and Al oxides in soils of the central Amazon may produce a similar effect, and thus changes in internal fluxes resulting from slash-and-burn agricultural practices in the tropics may not be evident from solute budgets or stream chemistry alone.

Intra-system processes and their effects on solute evolution can be evaluated by measuring the solution chemistry of different stages of a hydrologic pathway. Studies of chemical transformations along hydrologic pathways in tropical catchments are rare (McClain et al. 1994), and the effects of slash-and-burn agriculture on solution chemistry have yet to be investigated in the central Amazon. Hydrological studies have been done in the Amazon basin (Nortcliff & Thornes 1978; Nortcliff et al. 1979; Nortcliff & Thornes 1981; Leopoldo et al. 1982; Lesack 1993a, b; Vegas-Vilarrúbia et al. 1994), but these were located in areas of intact rain forest and do not address the effects of deforestation. Research concerning elemental dynamics associated with deforestation in tropical areas remains sparse (Uhl et al. 1982; Hamilton & King 1983; Uhl & Jordan 1984; Parker 1985; Jordan 1987).

The objective of our study was to document the effects of slash-and-burn agriculture on the solution chemistry at different stages along a hydrologic pathway in a small tropical rainforest catchment. The stages in the hydrologic pathway included precipitation, overland flow, shallow throughflow (herein defined as soil water percolating vertically through upper soil horizons), groundwater, bank seepage and stream water. Comparisons of solute concentrations at different stages were used to infer the intra-system processes that affect solute evolution in upland catchments (*terra firme*) of the central Amazon, with particular emphasis on those which affect nitrogen. The magnitude of the effects of deforestation was estimated by comparing solution chemistry from slash-and-burn areas to their pre-cut baselines, as well as to adjacent uncut controls.

Study area

Lake Calado is located in the central Amazon basin (3°15′ S, 60°34′ W) on the north side of the Solimões River (Amazon main stem) about 80 km W of Manaus (Figure 1). This region has distinct dry (June to November) and wet seasons (December to May; Lesack & Melack 1991; Williams et al. 1997). Although much of the Lake Calado basin was in various stages of development at the time of this study, most settlement of the study site area occurred after 1987 when an access road was constructed joining Lake Calado to the main road between Manaus and Manacapuru (Figure 1).

The study site was a sub-basin (23.4 ha) situated within the upland drainage of Lake Calado. The experimental hillslope area, slated for burning by one of the local residents, consisted of about 2 ha immediately downstream from an adjacent control area of about 5 ha bordering a small, first-order stream named the Braço do Mota, and is located at the head of the Igarapé do Mota (Figure 2). The area is hilly with elevation ranging from 15 to 25 m (about 100 m a.s.l.) and slopes ranging from 3 to 40%. Soils are classified as Oxisols consisting of a layer of sandy clay loam (0–50cm), charcoal and laterites underlain by a deep layer of loamy sands of various hues (Williams 1993). Above the mineral soil is a mat of humus and roots up to 20 cm deep. A riparian buffer strip about 15 m wide was left uncut by the landowner, and there was a small section (about 0.5 ha) of dense secondary forest in the experimental plot before cutting (Figure 2).

The forest of the partially deforested catchment was completely intact and mature *terra firme* in an aerial photograph of the Lake Calado basin in 1955, with some selective cutting of the lower reaches (downstream of the

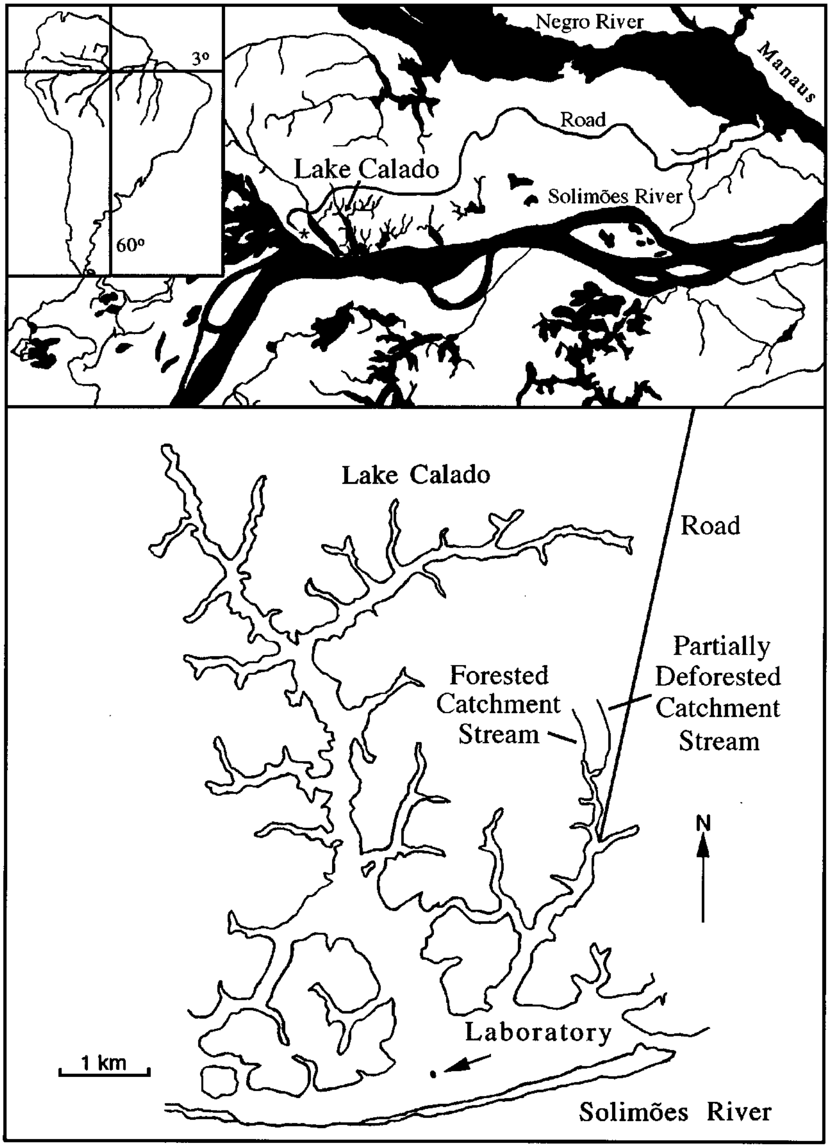


Figure 1. Map showing the location of the study site within the Lake Calado drainage basin. Manacapuru (west of Lake Calado) is indicated by an asterisk.

experimental and control plots) prior to 1977. Lesack (1993b) states that the forest of the study site consisted of 40- to 50-year old stands physiognomically similar to mature forest except for the lack of emergents and large lianas. In 1989, Forested areas of the study site catchment ranged physiognomically

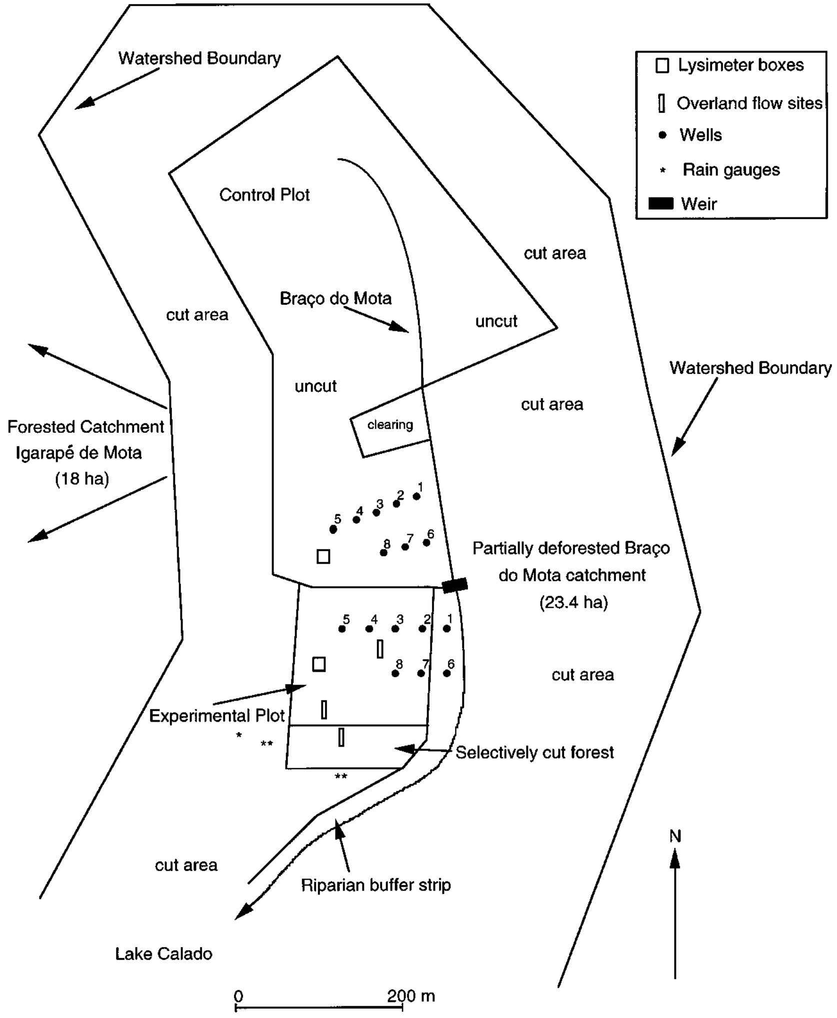


Figure 2a. Study site with equipment locations. The swath designated as the selectively cut forest was cut and burned at the same time as the experimental plot forest. The forested Igarapé de Mota (18 ha) is directly to the west of the partially deforested Braço do Mota catchment.

from low secondary successional vegetation (1–10 m) to aggrading forest 15–25 m high. No between-plot differences in vegetation were apparent from survey data obtained before the experiment began.

Species recorded during the vegetation survey of the forested area within the partially deforested catchment boundary are given in Williams (1993), and are briefly summarized here. Riparian vegetation located in the lower stream reaches had dense thickets of pioneer species such as *Bactris spp., Psychotria*

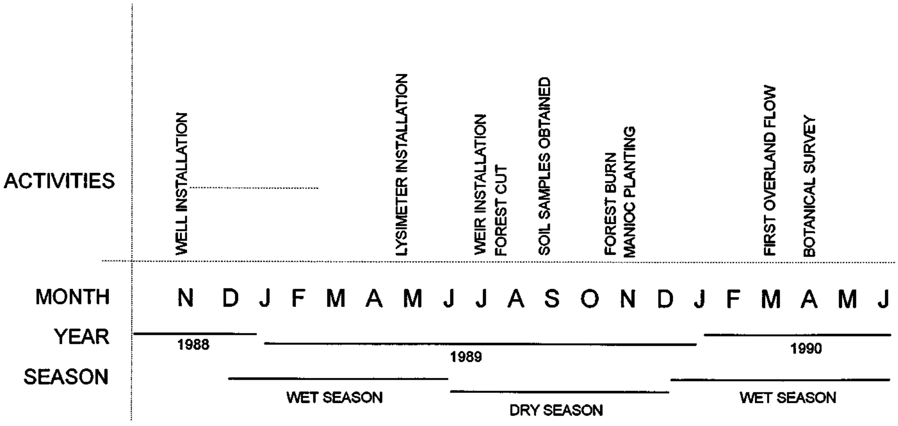


Figure 2b. Time line indicating the important events and equipment installations that occurred during the study.

spp., *Miconia spp.* and *Goupia glabra*. *Cecropia leucocoma* was common in secondary forest and the riparian zone at the study site. The forested upper reaches were well shaded by mature forest species with individuals from families such as Euphorbiaceae, Lecythidaceae and Melastomataceae, and the understory had large numbers of lianas such as *Doliocarpus spp.* and small palms such as *Euterpe precatoria*.

Hydrochemical changes caused by cutting and burning in the study site catchment were measured. Solute concentrations were analyzed in wet deposition, overland flow, shallow throughflow, groundwater and bank seepage in the experimental and control plots, and in stream water in the partially deforested and forested catchments. The experimental plot was cut in July 1989, burned in October 1989, and planted to manioc several weeks after burning (Figure 2b). Approximately 60% of the ground surface in the experimental plot was ashed as a result of burning.

Methods

Rain. Quantitative measurements of rainfall were made both at a floating laboratory and at the study-site catchment (Figure 1). Paired rain gauges (10 cm orifice) and an automatic tipping bucket rain gauge (15 cm orifice) were used for the collection of rain quantity at the catchment site. A complete description of rain chemistry can be found in Williams et al. (1997); data presented here are used for comparison with the solute composition of other pathway stages.

Overland flow. Overland flow was collected in plastic gutters (1 m long) mounted flush with the ground surface and perpendicular to the slope (Williams 1993). Collectors were rinsed with deionized water (DIW) approximately three times per week, and only samples collected within several hours after a rain event were analyzed. Three overland flow collectors were installed in the experimental plot. Overland flow did not occur in forested areas of the study catchment at any time during several of the largest storms recorded (pers. obs.). Consequently, overland flow collectors were not installed in the control plot, and we assume that there was no overland flow at any time during the period of study in the control plot.

Shallow throughflow. Throughflow was collected with zero-tension lysimeters in the control and the experimental plots. Wooden shelters built to protect the lysimeter end caps and collection bottles were constructed with detachable roofs and placed at ground level in 1 m^3 pits dug into the soil surface of both plots (Staver et al. 1987). Installation began approximately 5 months before the first sample collection to allow sufficient time for intimate soil contact with the lysimeters to develop and reduce disturbance effects (Shepard et al. 1990); four lysimeters were installed in each 1 m^3 pit (one per plot) with one lysimeter extending from each wall. Lysimeters were positioned horizontally at the bottom of the rooted zone 50 cm below ground surface in both pits. The lysimeters consisted of two 75 cm lengths of polyvinyl chloride (PVC) joined together by a flush, watertight male/female thread; one side of the outer section was slotted (0.5 mm) and the slots positioned upward, extending from about 50 to 125 cm from the edge of the wooden shelter (Williams 1993). Each lysimeter sample was analyzed separately, and the values were pooled to represent an infiltration event. When sample volume was limited, aliquots from a single pit were combined for analysis. Lysimeter samples were not volume-weighted.

Estimates of flux rates of water were obtained from both plots during the month of March 1990. This was accomplished by using small, soil-filled funnels (10 cm inner diameter) plugged with a ball of glass fiber and connected to a collection bottle by tygon tubing (one per plot). The funnels were braced against the ceiling of a small cavern located 50 cm below the soil surface at the same depth as the lysimeters. Water volumes were measured about 2 hours after storm events using a graduated cylinder. Soils in the vicinity of the funnels were structurally similar and without obstructions such as large roots or rocks. Estimates of water flux were used for between-plot comparison only, and no spatial or temporal extrapolation was done due to the tendency to underestimate soilwater fluxes (Sollins 1985).

Groundwater. Groundwater was measured in 16 wells that were arranged in replicated transects in both the experimental and the control plots (8 per plot, Figure 2). Wells ranged from 1 to 16 m in depth and were spaced at 15 m intervals upslope from the primary wells located 2 m from the edge of the stream. The experimental and control plots had two well transects each, one transect with 5 and the other with 3 wells. The top two wells in the second transect of both plots were not replicated. The top-slope (#5) and the streambank wells (#1 and 6) are not included in between-plot comparisons because solute concentrations in these wells remained relatively invariant after cutting and burning.

Wells were manually drilled to the water table with a bucket type soil auger and then with a sand auger to drill below the water-saturated soil zone. Iron concentrations were initially very high in all wells, apparently due to drilling the wells with an iron auger, and Fe data were divided into subsets (before and after December, 1989) to eliminate the effects. All wells were fitted with a plastic collar about 5 cm in width that was placed 50 cm below ground level to prevent water and soil movement down the outside shaft of the well. Well screens 1 m in length with 0.5 mm slots were fitted to the end of the well shafts. All well screens were soaked in 30% HCl for 1 hour, rinsed and leached in DIW overnight. Well installation was completed 6 to 8 months prior to cutting of the forest in the experimental plot (Figure 2b).

Water table levels were measured using a marked length of tygon tubing with a small lead weight on the end. This was used as a sounding line by listening for bubbling when the tube made contact with the water surface as air was blown through the tube. The water table was recorded, and the wells were purged of water 1 day before sample collection. Water samples were collected with a PVC bailer attached to a spool of nylon fishing line and returned to the lab in pre-rinsed polyethylene bottles used specifically for each well.

Seepage meters. Seepage meters (Lee & Cherry 1979) were used periodically to measure the water chemistry of bank seepage along the length of the stream. Devices were constructed from the top portion of a polycarbonate water jug which was inserted into the stream bank leaving only a small opening through which water passed into a collection bag. Seepage meters were allowed to flush over 24 hours to remove water trapped during installation, and then collection bags were mounted on the devices. Seepage-meter samples were collected at most two days after installation to reduce chemical changes during the collection interval. In addition to the measurements described above, groundwater was collected with bottles placed under small springs emerging on the hillslope up to 1 m away from the stream. These streambank measurements are referred to as spring-seepage samples.

Stream water. Stream discharge was measured with a sharp-crested 120° V-notch weir and stilling well water level recorder. An adjacent uncut catchment stream named the Igarapé de Mota (18 ha) served as a control for streamwater chemistry (Figure 2). A complete description of stream chemistry can be found in Williams & Melack (1997); data presented here are used for comparison with rain and soil waters.

Soils. Descriptive soil profiles over the depth range of 1 to 16 m were recorded during the augering of eight wells in both plots (Williams 1993). Additionally, soil samples from depths ranging from 35 to 100 cm were collected with a bucket-type soil auger in four locations (one core at each location), and core sections were separated according to visible changes in soil color and texture. Samples were obtained from the bottom and top of the hillslope in both the experimental and control plots, and soil parameters measured from forested areas were compared with those from the upper slope of the deforested plot. Soil samples were placed in polyethylene bottles, refrigerated, and sent back to the United States for the analysis of *p*H, available nitrogen, exchangeable cations, exchangeable acidity and cation exchange capacity (CEC). All soil analyses were performed according to protocols of Page et al. (1982).

Equipment installation and sample collection. A time line of the equipment installations and catchment disturbances is given in Figure 2b. The landowner cut the experimental forest in mid-July 1989. Trees were felled by ax and ground disturbances were minimal. The forest was burned in mid-October 1989, and some of the unburned wood was removed from the experimental plot before planting manioc in early November 1989.

The frequency of sample collection was different for each stage of the hydrologic pathway that was sampled. Rain was collected on an event basis whenever possible (Williams et al. 1997). Lysimeter and overland flow samples were collected within 12 hours after a rain event occurred and promptly returned to the lab for analysis. Well samples were collected monthly, but weekly to biweekly during the initial 3 months after cutting the experimental plot forest. Seepage-meter samples were taken opportunistically, and stream water was collected at least once per week.

Two important factors in interpreting our chemical data are the various samples sizes of different pathway stages and the heterogeneity of the forest burn. Rainfall, groundwater and stream water have samples sizes of about 115, 150 and up to 60, respectively, whereas overland flow, throughflow, bank seepage and soils have sample sizes <20. Although the smaller data sets are used for comparative purposes among pathway stages and not, for instance, for calculating solute fluxes, their representativeness should be viewed with caution.

Secondly, due to an extended period of rainfall in early October 1989 prior to burning, the cut forest did not burn completely. Although felled trees were burned, boles remained and about 40% of the ground surface in the experimental plot was unashed, including the lysimeter area. Thus, although partial burning may commonly occur in agricultural plots of the central Amazon, the soil and throughflow data presented here are not representative of a slash-and-burn disturbance; rather, the data are indicative of changes which occur after a cut-and-fallow disturbance followed by planting. However, overland flow, groundwater, bank seepage and stream water incorporate the effects of burning. Planting occurred two weeks after the burn by digging a hole in the topsoil (about 20 cm deep) for each manioc stem to be planted, but further disturbance was minimal since harvesting took place after the end of our measurements. No fertilizer was applied.

Chemical analyses. The same suite of handling and analytical procedures were performed on water samples collected from each pathway stage. A deionizing system provided DIW with a conductivity below 1 μS cm^{-1}, and rinse-water blanks showed no contamination. We used a plastic Nuclepore filtration apparatus and Gelman A/E glass fiber filters (1 μm nominal pore size), which were pre-rinsed with 200 mL of DIW. Filtered aliquots were placed into new high density polyethylene bottles that had been leached for 24 hours and then rinsed 3 times with DIW. All filtered samples were then refrigerated at about 4 °C until analysis. Measurements of pH were standardized to 25 °C, and statistics were calculated using [H$^+$]. Analytical methods and quality controls are summarized in Williams et al. (1997) and Williams & Melack (1997).

Data analysis. Volume-weighted mean (VWM) concentrations of all solutes in precipitation and stream water were calculated as follows:

$$VWM = \frac{\sum_{i=1}^{n} C_i V_i}{\sum_{i=1}^{n} V_i}$$

where, C_i is the concentration of a species in sample *i*, V_i is the volume of precipitation or discharge associated with sample *i*, and *n* is the total number of samples for which measurement was made. Arithmetic means of concentration were calculated for overland flow, throughflow, groundwater and bank seepage.

Results

Overland flow. Overland flow occurred in the experimental plot 5 months after burning, and apparently did not occur in the control plot. A 2-sample *t*-test showed that most solute concentrations in overland flow were significantly higher than in rain. pH was much higher in overland flow than in rain and ranged from 5.5 to 9.0 ($n = 14$), with the highest values occurring in the initial post-burn period. The P fractions increased proportionally more in concentration than the N fractions in transition from rain to overland flow (Table 1). Potassium in overland flow was lowest in concentration among the base cations, but had the largest relative increase over concentrations measured in rain. The concentration of dissolved organic carbon (DOC) was more than a factor of 5 times higher in overland flow compared to the rain which caused the flow (Table 1).

Hydrochemical losses from the rooted zone. Significantly greater vertical flows of water were measured in the experimental plot than in the control plot ($n = 15$; Figure 3). The ratio of rainfall infiltration to event volume (cm throughflow/cm rain) ranged from 0 to 0.59 in the control plot and 0.19 to 0.80 in the experimental plot. Smaller storms resulted in proportionally smaller infiltration rates, probably due to evapotranspiration and unsaturated water holding capacity.

Throughflow chemistry was determined from 4 pre-cut and 10 post-cut rain events yielding adequate volumes to perform analyses. Throughflow samples collected before cutting were not significantly different between plots for the N and P fractions, major ions and SiO_2. Disssolved inorganic carbon (DIC), DOC, organic acids and trace metals were not measured in these pre-cut samples.

Most solute concentrations of throughflow in the experimental plot had large initial increases after cutting and higher concentrations than in the control (Figure 4); solutes generally showed post-cut declines in concentration after the initial increase (Figure 5). Nitrate, Na^+, Cl^- and K^+ were leached from the rooted zone in relatively large quantities, and NO_3^-, SO_4^{2-}, K^+ and NH_4^+ had the largest changes in concentration after burning. Concentrations of most solutes in throughflow were significantly greater in the experimental plot compared to the control using a 2-sample *t*-test (Table 1). Nitrate increased from a pre-cut baseline of 120 to over 1800 μeq L^{-1} at the beginning of the wet season in December 1989, decreasing exponentially to 500 μeq L^{-1} by the end of May 1990 (Figure 5); $[NO_3^-]$ in the control plot was higher also than its pre-cut baseline. The base cations, Cl^-, SO_4^{2-} and trace metals followed trends similar to NO_3^-. DOC concentrations in throughflow

Table 1. Volume-weighted mean concentrations for solutes in rain (RN) and stream water (SW); mean solute concentrations for shallow throughflow (ST), groundwater (GW), overland flow (OF), and bank seepage (BS). Deforested and forested plots are designated with a D or F, respectively. Bank seepage is designated with a S (spring seepage) or M (seepage meter), respectively. Concentrations were determined from samples collected from July 1989 to July 1990. All units are μM, except ANC (μeq L^{-1}), pH and Mn (nM). Unfiltered samples are designated by "U". Asterisks indicate significant differences ($p < 0.05$) between comparisons of rain and overland flow, deforested and forested sites, and spring and seepage meter samples.

		RN	OF	ST		GW		BS		SW	
Solute				D	F	D	F	S	M	D	F
NH_4^+		3.0	7.6*	4.4*	0.7	20.6*	6.9	0.4*	6.8	1.0*	0.4
NO_3^-		4.2	7.5*	806*	210	68.0*	24.5	35.0*	2.8	11.6*	8.5
TDN		6.8	43.9*	–	–	92.2*	43.5	37.0*	24.0	19.9*	13.1
TN	U	10.8	–	–	–	–	–	–	–	24.3*	15.8
PO_3^{3-}		0.03	1.37*	0.05	0.6	0.17*	0.09	0.12	0.12	0.11	0.07
TDP		0.25	3.05*	0.35	0.56	0.36*	0.32	0.21	0.21	0.46*	0.29
TP		0.41	–	–	–	–	–	–	–	0.56*	0.36
Na^+		2.4	40.4*	191.0*	19.1	61.0*	26.5	31.7	24.8	15.6*	11.1
K^+		0.8	26.0*	58.4*	1.1	10.9*	4.4	3.4	2.3	5.4*	1.5
Ca^{2+}		1.2	26.2*	5.6*	1.4	11.1*	4.5	12.1	12.2	4.3*	2.7
Mg^{2+}		0.5	14.2*	7.2*	0.8	1.8*	0.5	6.5	6.0	2.9*	1.2
Cl^-		4.6	23.8*	98.8*	11.9	19.5	23.2	15.3	20.4	13.8	11.4
SO_4^{2-}		1.0	31.0*	17.1*	2.2	2.6*	0.7	21.3	25.6	5.7	1.8
SiO_2		0.0	19.1*	55.6	41.1	53.5	57.6	62.0*	82.6	71.6*	90.2
H^+	U	17.0	1.3*	195.3	140.7	16.4	38.8	14.0	16.0	11.2*	18.4
ANC	U	−5.0	193.0*	–	–	117.0*	36.5	−2.0	25.0	3.4*	−8.1
pH	U	4.7	5.9*	3.7	3.9	4.8	4.7	4.9	4.8	5.0*	4.7
μS cm^{-1}	U	6.5	24.4*	192.2*	45.5	28.0*	16.2	12.7	10.2	8.4	9.3
DOC		159	869*	194*	121	138*	54	160*	421	158*	73
DIC		0	–	–	–	1.43	1.24	–	–	0.42	0.44
Al		0.3	2.3*	130.5*	29.6	1.8	1.5	1.7	1.5	1.5	1.9
	U	0.3	8.1*	–	–	–	–	–	–	4.2	3.6
Fe		0.1	0.7*	0.4*	0.2	1.0	0.7	1.0	1.4	0.7	0.5
	U	0.1	3.9*	–	–	–	–	–	–	3.0	1.5
Mn		7	15*	637*	36	182	164	36	25	55	36
	U	6	73*	–	–	–	–	–	–	36	36

in the experimental plot were similar to those measured in the rain that caused the infiltration (about 200 μM), whereas those of the control plot were significantly lower (Figure 4).

Changes in groundwater chemistry. Groundwater levels above stream height were not significantly different between plots using a paired *t*-test. Ground-

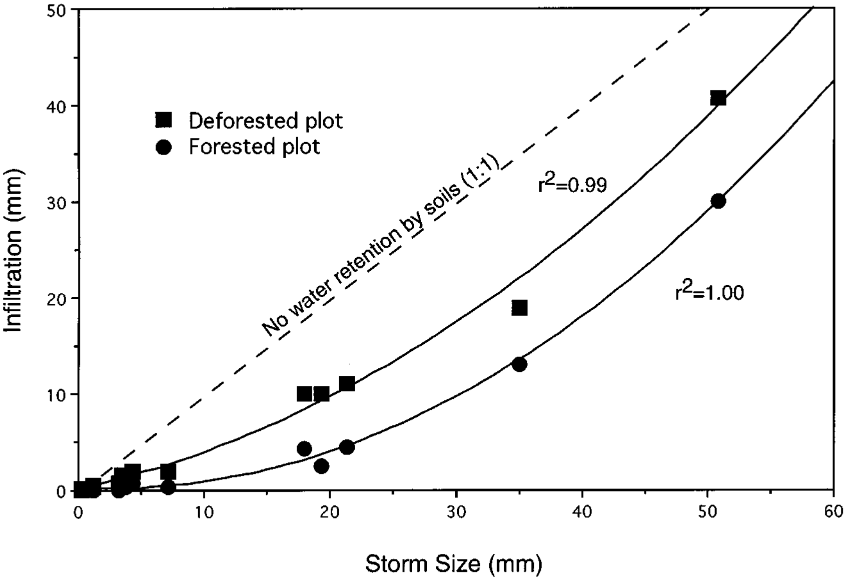


Figure 3. Soilwater infiltration volumes measured in the experimental and control plots during the period of study.

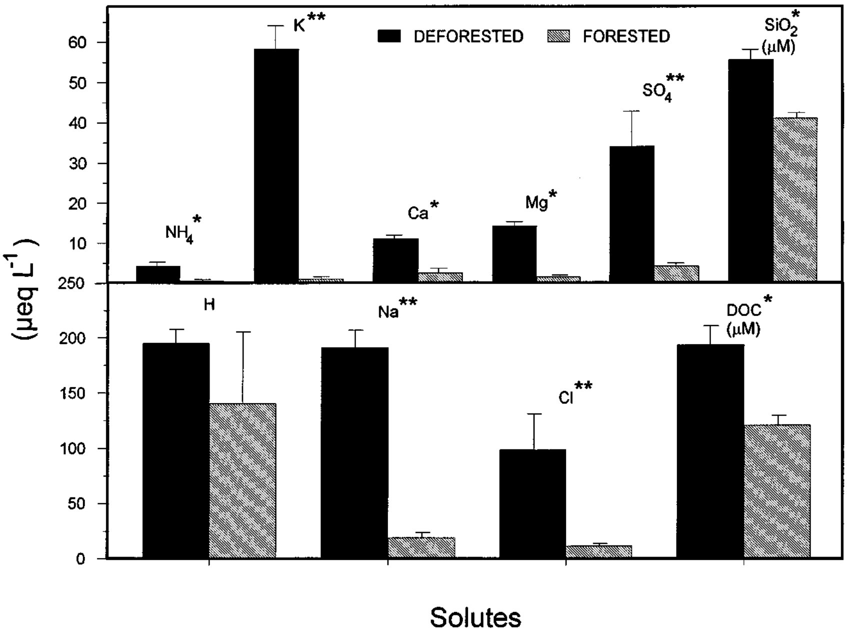


Figure 4. Mean solute concentrations for shallow throughflow in the experimental and control plots during the period of study. Levels of significance are designated by asterisks (i.e. * when $0.05 > p > 0.01$ and ** when $p < 0.01$). Error bars represent standard errors.

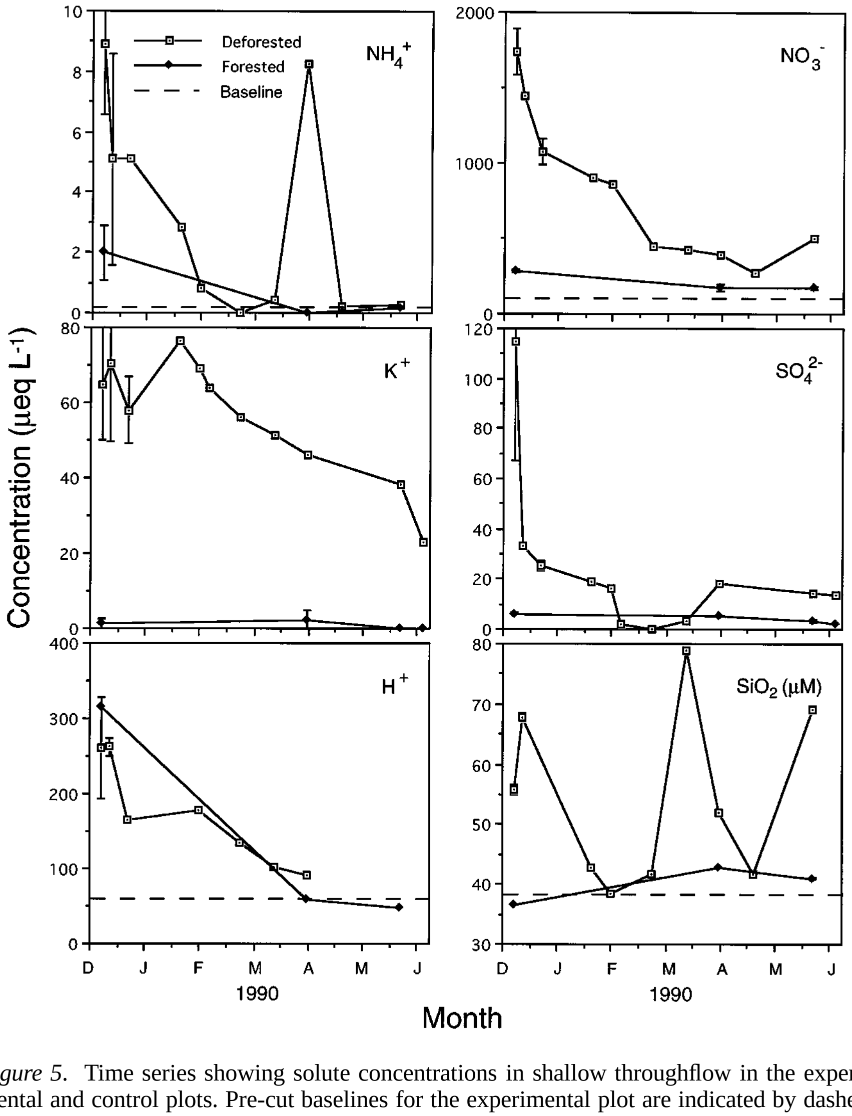


Figure 5. Time series showing solute concentrations in shallow throughflow in the experimental and control plots. Pre-cut baselines for the experimental plot are indicated by dashed lines. Error bars represent standard errors.

water levels generally increased in January after the beginning of the wet season in December and peaked in late May just after peak rainfall (Figure 6). The magnitude of vertical watertable fluctuations generally increased from lower (~0.5 m) to upper (~1.5 m) hillslope well sites in both plots. The hydraulic gradients calculated from the top well to the stream in the deforested plot were similar to those of the forested plot and ranged from 0.5 to 4 cm m^{-1} over the groundwater hydrograph.

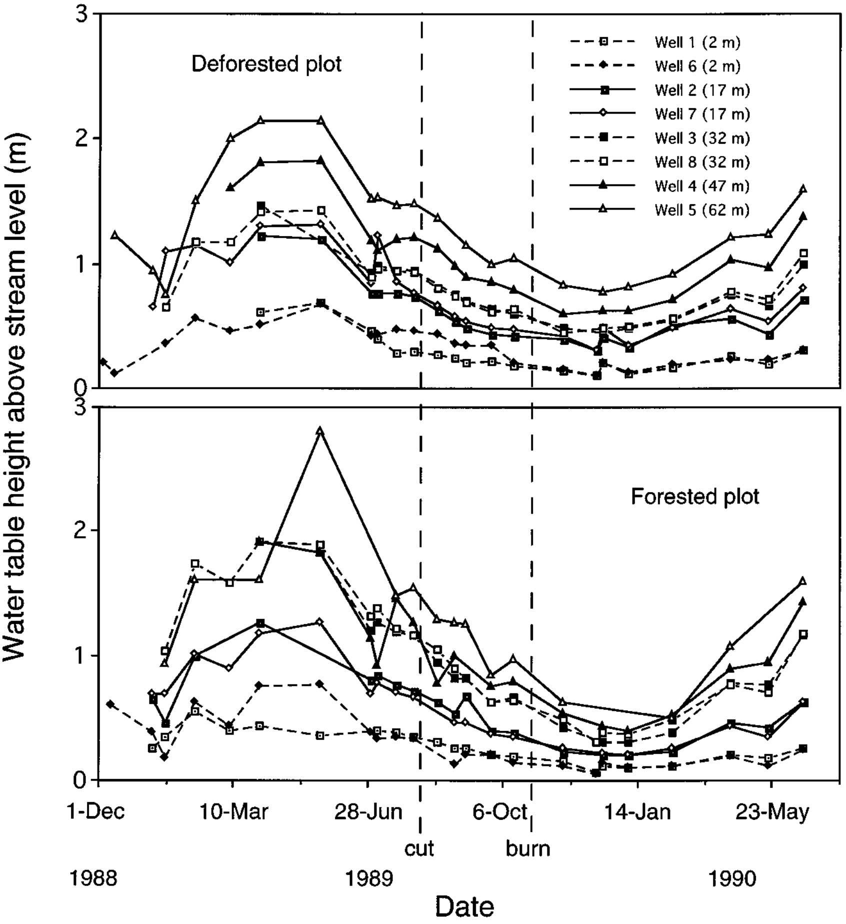


Figure 6. Water table fluctuations relative to stream level recorded in the experimental and control plots from December 1988 to May 1990.

Spatial variability of solute concentrations within each plot was high. Solute concentrations were lower in wells at the top and bottom edges of the hillslope compared to centrally located wells for all solutes except DOC, which was highest in the top well (#5) of the experimental plot during the first 5 months after cutting (384±66 μM, $n = 8$) compared to the mid-slope average of 194 μM (Table 1). Mean [NO_3^-] was about 68 in the mid-slope compared to 4 μM in the fringing wells (#1, 5 and 6) of the experimental plot. In order to separate temporal and spatial effects of cutting and burning in the experimental plot, we evaluated changes over time using only data from the mid-slope wells (#2, 3, 4, 7 and 8), because these wells were affected most by the disturbance.

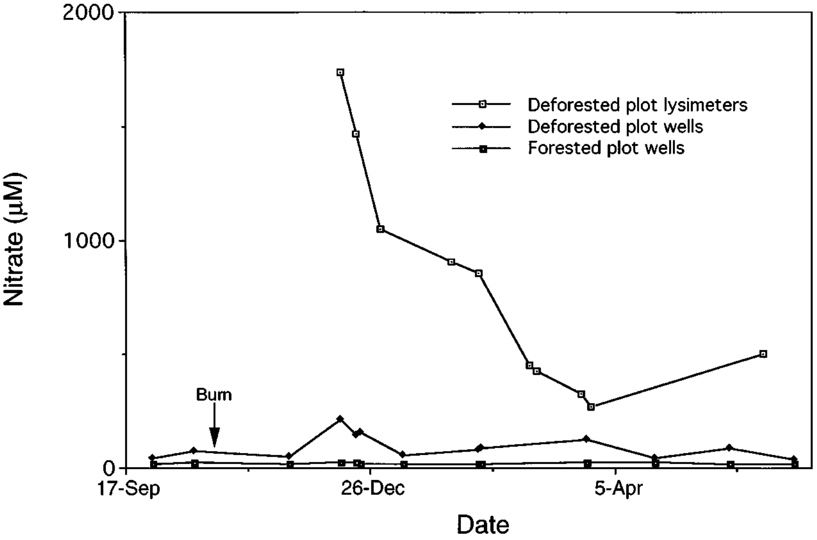


Figure 7. Concentrations of NO_3^- in the shallow throughflow (lysimeters) of the experimental plot, and groundwater NO_3^- in the mid-slope wells of the experimental and control plots, June 1989 to May 1990.

The wells of the experimental plot had higher concentrations of NO_3^- compared to the pre-cut baseline possibly due to high NO_3^- throughflow percolating to the water table at the onset of the wet season (Figure 7). A comparison of $[NO_3^-]$ in the mid-slope wells of the experimental plot to the pre-cut baseline showed 14-fold increases after cutting and 57-fold increases after burning (Figure 8). In contrast, $[NO_3^-]$ in the control-plot wells increased slightly to 23.7 μM compared to a pre-cut baseline of 15.6 μM (n = 37, SE = 2.7), despite $[NO_3^-]$ in throughflow that was occasionally larger by a factor of 10.

Groundwater was mildly acidic in both plots (pH = 4.8) and solute concentrations were moderate relative to the other pathway stages (Table 1). Compared to throughflow, concentrations of most solutes were significantly lower in experimental plot groundwater using a 2-sample t-test. A similar comparison for the control plot showed that the concentrations of about half of all the solutes were significantly higher in groundwater than in throughflow. Between-plot concentration differences of groundwater using a paired t-test showed that most solutes were significantly higher in the experimental plot compared to the control (Table 1).

Large changes in acid neutralizing capacity (ANC) occurred in the wells of both plots (Figure 9). Groundwater ANC of the experimental plot was consistently higher than that of the control plot, although the largest difference

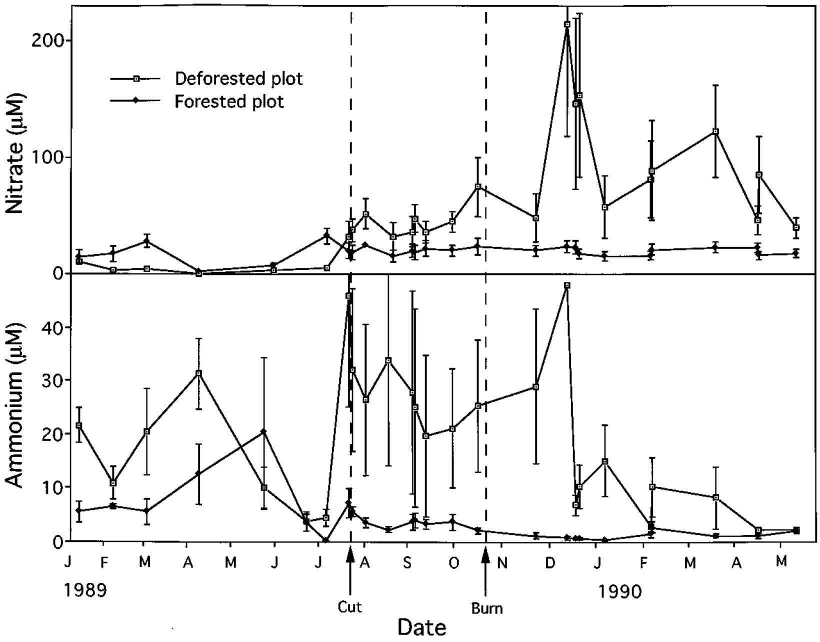


Figure 8. Mean NO_3^- and NH_4^+ concentrations observed in the mid-slope wells of the experimental and control plots, January 1989 to May 1990. Error bars represent standard errors.

predated the cut. The contribution of HCO_3^- (pK = 6.3) to total titration alkalinity was very small given that pH values of groundwater were commonly <5. Consequently, [DIC] in groundwater was <2 μM. Hydrogen ion in the experimental and control plots was similar, except for a 2-month period immediately after the burn when H^+ was lower in the deforested plot (Figure 10).

Groundwater [DOC] in the experimental plot was higher than in the control plot throughout the sampling period (Figure 10). DOC concentrations in the plots largely fluctuated in parallel, but deforested-plot DOC concentrations were higher and diverged from forested values after the burning of the experimental plot in October 1989. [DOC] and [NO_3^-] paralleled one another throughout the sampling period in both plots (compare Figures 8 & 10). The time series of the remaining major solutes in groundwater show that most of the divergence in concentration between plots occurred after the burn (Figure 9).

Comparison of bank seepage. Ammonium and NO_3^- concentrations in samples collected using seepage meters in the upstream control area were

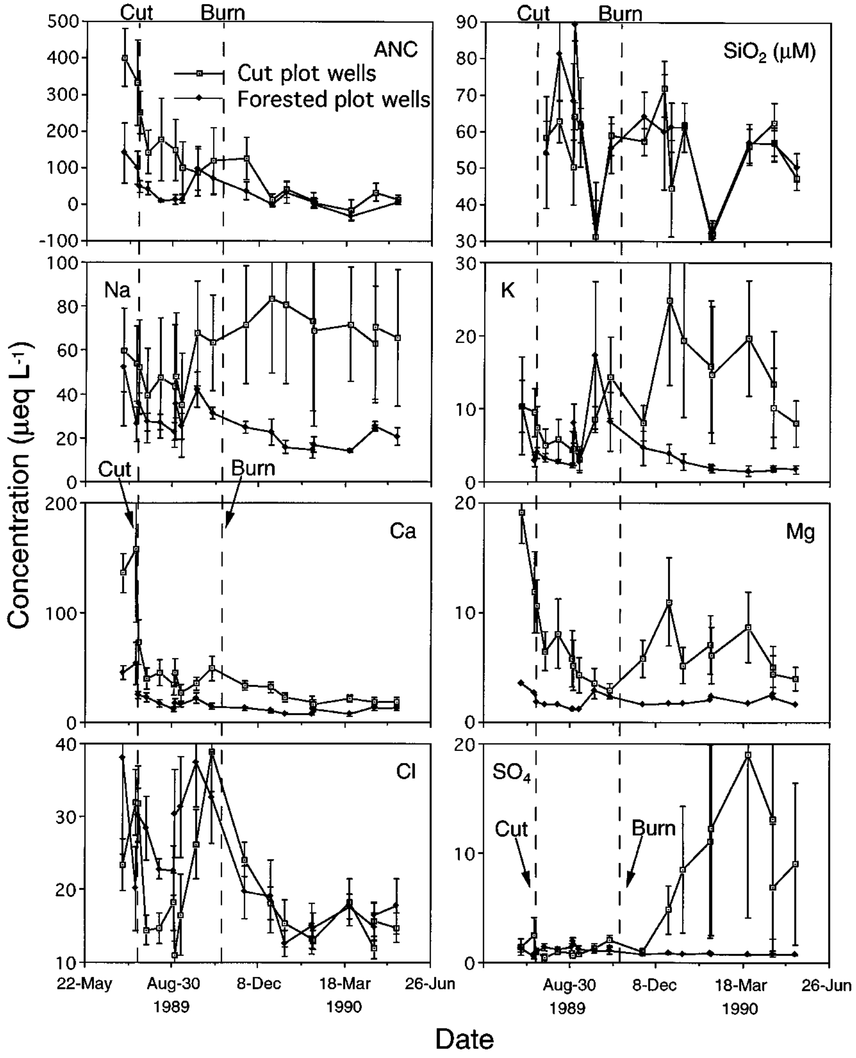


Figure 9. Time series of concentrations for selected solutes in the mid-slope wells of the experimental and control plots, July 1989 to May 1990. Error bars represent standard errors.

slightly larger than those from the downstream experimental area (means of 13.9±0.8 versus 7.9±4.7 μeq L^{-1}; respectively). Spring seepage in cut areas had significantly higher concentrations of NO_3^- and total dissolved nitrogen (TDN) than seepage-meter samples, but lower NH_4^+, SiO_2 and ANC (Table 1). Phosphorus concentrations were the same for both types of seepage samples. [DOC] in the seepage-meter samples was significantly greater than in that of the streambank springs, and most other solutes were similar in concentration.

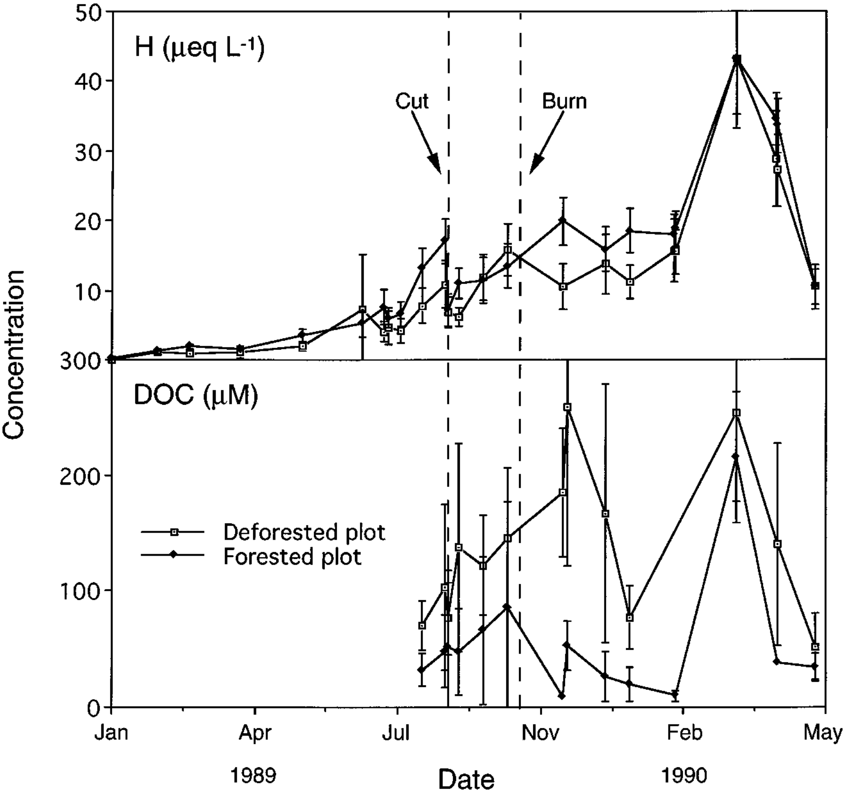


Figure 10. Mean groundwater DOC and H^+ concentrations for the experimental and control plots (mid-slope wells). Error bars represent standard errors.

Nitrogen evolution in the riparian zone. Transformations of NO_3^- and NH_4^+ occurred along the presumed pathway of groundwater to stream water (Figure 11). Nitrate was 2.8 μM in bank seepage collected with the seepage meters compared to 8.2 and 8.5 μM in the streambank wells and stream water, respectively. Conversely, $[NH_4^+]$ was 6.8 μM in the seepage-meter samples compared to 3.9 and 1.0 μM in the streambank wells and stream water, respectively.

Stream water. A detailed description of streamwater runoff and chemistry for forested and partially deforested catchments is given in Williams & Melack (1997). Streamwater chemistry is provided in Table 1 for comparison with the other pathway stages monitored in this study. Streamwater concentrations of the base cations, total and total dissolved phosphorus (TP and TDP, respectively) and DOC in the partially deforested catchment were higher than those

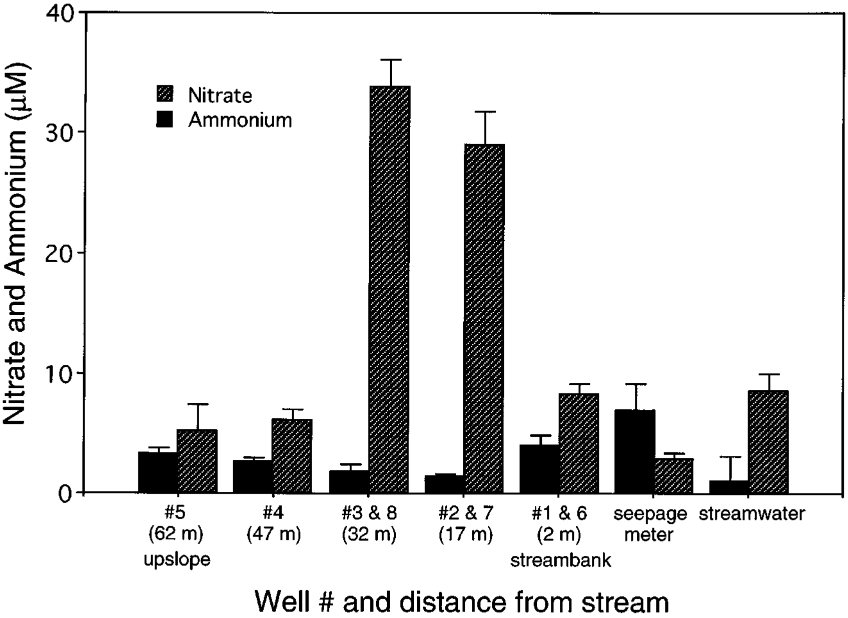
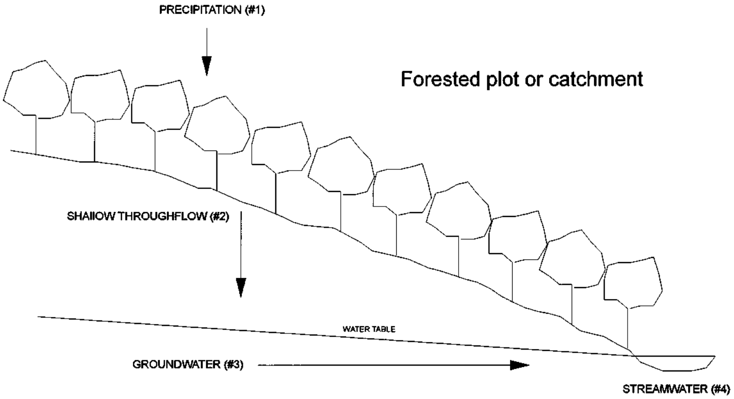


Figure 11. Comparison of NO_3^- and NH_4^+ in a transect from the upslope well (#5) to the stream in the forested plot. There were no reliable seepage data from deforested areas, making a similar comparison using transect data from the cut plot impossible. Error bars represent standard errors.

from the adjacent forested catchment, whereas H^+ and SiO_2 concentrations were lower.

Effects of cutting vegetation on soils. Soil samples obtained from shallow soil cores (<100 cm) one month after cutting had pH ranging from 4.2 to 5.4, and CEC values from 6 to 14 mmol $[100\ g]^{-1}$ (Table 2). Soils from forested areas had a mean pH of 4.9, and mean available N of 4.3, 3.4 and 0.04 $\mu g\ g^{-1}$ for NH_4^+, NO_3^- and NO_2, respectively. Concentrations of exchangeable cations in forested areas were <1.3 meq $[100\ g]^{-1}$ and CEC was 9.1 mmol $[100\ g]^{-1}$. Nitrate and H^+ concentrations were significantly higher in the deforested upper slope than in soils from forested areas.

Diagrammatic summaries. Solute retention or mobilization in the control and experimental plots were determined by comparing the solute concentrations of the different pathway stages. Significant ($p < 0.05$) retention was indicated by lower solute concentrations in a subsequent pathway stage than those of the preceding stage, whereas significant ($p < 0.05$) mobilization was indicated by the opposite scenario. In general, solute concentrations were higher in over-



	PO_4	TDP	TP	NH_4	NO_3	TDN	TN	Na	K	Ca	Mg	H	ANC	Cl	SO_4	Al	Fe	Mn	DOC	DIC	Si
#1 & #2	+	+	NA	-	+		NA	+				+		+	+	+	+	+	-	+	+
#2 & #3			NA	+	-		NA	+	+	+	-	-	+	+	-	-		+	-	+	+
#3 & #4			NA	-	-	-	NA	-	-	-	+		-	-	+			-		-	+
#1 & #4	+			-	+	+	+	+	+	+	+			+	+	+	+	+	-	+	+

Figure 12. Diagrammatic summary of solute mobilization along the hydrological pathway stages monitored in the control plot. The first comparison is between precipitation (#1) and shallow throughflow (#2) in the control plot. No overland flow was observed in the control plot. Other comparisons are between shallow throughflow (#2) and groundwater (#3), groundwater (#3) and stream water (#4), and rain (#1) and stream water (#4) in the forested plot. All tests of significance were done using a 2-sample *t*-test. Significantly higher and lower ($p < 0.05$) concentrations compared to a previous stage are indicated by "$+$" and "$-$", that represent mobilization or retention of solutes, respectively. "NA" indicates that data were not available, and a blank indicates that there was no significant difference between pathway stages.

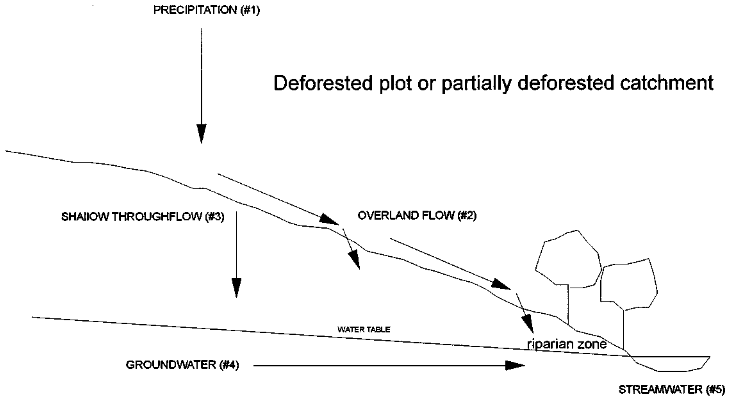
land flow and throughflow compared to rain in both plots (Figures 12 & 13), although the magnitude of the mobilization was higher in the experimental plot (Table 1). In the control plot, mobilization of most solutes from throughflow to groundwater occurred. In contrast, solutes were generally retained or diluted from throughflow to groundwater in the experimental plot. Solute concentrations in groundwater were generally higher in the experimental plot compared to pre-cut baselines and groundwater in the control plot wells. In a comparison of rainfall and stream water, mobilization of most solutes occurred in both catchments, although net export was higher in the partially deforested catchment.

Discussion

Overland flow chemistry. Overland flow was observed in the experimental plot about 5 months after burning in areas where substantial disintegration

Table 2. Results of soil chemistry derived from four shallow vertical profiles in both the experimental and control plots one month after cutting. “N” indicates that a Nuclepore 1.0 μm filter was used after primary filtration with a Gelman A/E glass fiber filter. “A” indicates that only Gelman filtration was performed. All exchangeable cation analyses were performed after filtration with Nuclepore 1.0 μm filters. Cation exchange capacity (CEC) is the sum of exchangeable cations and exchangeable acidity. Exchangeable acidity was determined by the KCl method in Page et al. (1982). All locations were forested except for the Upper Slope Deforested Plot site; the Lower Slope Deforested Plot site was located in the riparian forest area.

					Avail. N (μg g^{-1}) KCl extract, 2N 1:12			Exchang. cations (meq [100 g]$^{-1}$) in NH_4OAc(1:25)					
Site	Date	cm	pH sat. paste	Filter	NH_4-N	NO_3-N	NO_2-N	Ca	Mg	Na	K	Exchang. acidity	CEC mmol [100 g]$^{-1}$
Upper	(82289)	20	4.7	A	8.2	0	0.1	0.2	0.1	0.2	0.1		
Slope		30	4.2	A	2.8	16.3	0.1	1.1	0.3	3.2	0.5	8.6	13.7
Deforested		43	4.4	A	2.0	14.1	0	0.2	0.1	0.1	0.1		
Plot		55	4.6	N	1.2	8.5	0	0.1	0.1	0.1	0.1	5.7	6.1
Lower	(82289)	73	4.7	N	2.3	7.7	0	0.2	0.1	0.2	0.1		
Slope		12	5.0	A	4.9	1.1	0	0.5	0.1	0.1	0.1		
Deforested		22	4.9	A	2.6	5.4	0	0.4	0.1	0.1	0.1		
Plot		35	5.4	A	2.6	1.2	0.1	1.3	0.1	0.2	0.1		
Upper	(81889)	30	4.8	A	10	2.7	0	0.2	0.1	0.1	0.1		
Slope		50	4.6	A	3.8	5.8	0	0.5	0.1	0.1	0.1		
Forested		70	4.8	N	2.6	6.0	0.2	0.2	0.1	0.1	0.1		
Plot		90	5.1	N	3.0	3.1	0	0.9	0.1	0.5	0.1		
Lower	(82389)	100	4.9	N	3.2	1.9	0.1	0.8	0.1	0.2	0.1		
Slope		18	4.5	A	3.6	7.2	0	0.2	0.1	0.1	0.1	8.6	9.1
Forested		45	5.1	A	4.8	2.5	0	0.2	0.1	0.1	0.1		
Plot		80	5.1	N	6.6	0	0	0.3	0.1	0.1	0.1		



	PO_4	TDP	TP	NH_4	NO_3	TDN	TN	Na	K	Ca	Mg	H	ANC	Cl	SO_4	Al	Fe	Mn	DOC	DIC	Si
#1 & #2	+	+	+	+	+	+	+	+	+	+	+	-	+	+	+	+	+	+	+	+	+
#3			NA	+	+		NA	+	+	+	+			+	+	+	+	+	+		
#4	+	+	NA	+	+	+	NA	+	+	+	+		+		+				+		
#5		+	+	+	+	+	+	+	+	+	+	-	+						+		-

Figure 13. Diagrammatic summary of solute mobilization at different hydrological pathway stages in the experimental plot. The first comparison is between precipitation (#1) and overland flow (#2) of the experimental plot. The comparisons of shallow throughflow (#3) and groundwater (#4) are between the experimental plot and the control plot, while stream water (#5) is a comparison between the partially deforested catchment and the forested catchment. Between-plot differences in groundwater concentrations were tested for significance using a paired *t*-test, and all other tests of significance were done using a 2-sample *t*-test. Significantly higher and lower ($p < 0.05$) concentrations compared to a previous stage or a control are indicated by "+" and "−", that represent solute mobilization and retention, respectively. "NA" indicates that data were not available, and a blank indicates that there was no significant difference between pathway stages.

of the forest root mat had occurred. Overland flow in the experimental plot area always infiltrated into the soil before reaching the stream and was never significant enough to breach the 15 m buffer strip. Similarly, Hodnett et al. (1995) observed small-scale runoff at Fazenda Dimona, north of Manaus, but travel distances were short and most of the water ultimately infiltrated. Thus, although overland flow in the experimental plot transported soil particulates and solutes down slope, it was considered a transient pathway between rainfall and throughflow. However, buffer strips are rarely used along the edges of streams and lakes in the area, and older deforested plots in the Calado basin with steeper hillslope grades lacking buffer strips contributed substantial overland flow and eroded soil to stream and lake water during storm events (pers. obs.). Although data from older plots may not be representative of recent disturbances, it is important to recognize that overland flow probably contributes larger inputs of dissolved solutes and particulates directly to the lakes and streams of the region than this study would indicate.

Infiltration. Infiltration of water through the upper soil horizons and the concentrations of inorganic N in throughflow were higher in the experimental plot after cutting and burning compared to the control plot (Figures 3 & 5). Gaseous effluxes of N-oxides during burning were not measured in this study, but these can sometimes exceed post-fire leaching losses in temperate catchments (Grier 1975; Schoch & Binkley 1986; Lobert et al. 1991). Hence, both leaching and gaseous effluxes associated with fire can lead to a loss in soil fertility. The availability of inorganic N after burning is known to increase from the accelerated decomposition of the forest floor (Raison et al. 1990), and infiltrating rain water probably leached NO_3^-, which is particularly mobile under disturbed conditions (Vitousek et al. 1979; Vitousek 1980). In the temperate zone, disturbances generally decrease plant uptake of N, and increase decomposition and mineralization, which accentuate nitrification (Matson & Vitousek 1981; Vitousek & Matson 1985). For example, St. John & Rundel (1976) reported an increase in soil NO_3^- the first spring after burning a mixed-conifer forest, and Schoch & Binkley (1986) observed increased N mineralization rates in loblolly pine forests after burning. Chorover et al. (1994) observed higher A-horizon [NO_3^-] in soil solutions for over 2 years after the prescribed burning of a mixed-conifer forest in the Sierra Nevada, California. Increases in [NO_3^-] occurred in stream water after the cutting and herbicide treatment of a forested catchment at Hubbard Brook, and Likens et al. (1970) attributed the increase to nitrification.

The nitrification hypothesis in our study is supported by higher NH_4^+ and pH (the latter due to ashing) observed in overland flow (Table 1), both factors which promote nitrification. Moreover, the second and third strata of soil in the cut plot had lower pH and higher NO_3^- compared to the other 3 locations, which were forested (Table 2). Ammonium production probably continued after tree removal, and without the active uptake by tree roots (Stark & Jordan 1978), NH_4^+ can accumulate. Ammonium accumulation may occur concurrently in the control plot because the trees are probably P limited due to P adsorption by Al and Fe oxides.

Although less pronounced in the control, both plots showed seasonal removal of large amounts of NO_3^- in throughflow. The mineralization of leaf litter and organic matter stocks over the dry season may result in an accumulation of soluble materials such as inorganic N in the upper soil horizons of tropical catchments. Some of the N would then be mobilized from the soils during periods of high soil flushing, and the higher [NO_3^-] in the throughflow of both plots at the beginning of the wet season compared to the antecedent [NO_3^-] baseline support this hypothesis (Figure 5).

Incomplete burning in the experimental plot lysimeter area may have delayed the decomposition of the root mat and other organic matter stocks. The mineralization of unburned topsoils and their accompanying rootmats would then contribute progressively to the leaching process. For example, similar to NO_3^-, decreasing trends following initially high concentrations were observed for most solutes in throughflow (Figure 5). In contrast, completely burned areas commonly have large quantities of ashed organic matter stocks and soluble oxides which are converted to carbonates after burning, and solutes from this ash are generally known to increase soil pH and streamwater ANC (Tiedemann et al. 1978), although there are exceptions (Bayley et al. 1992b). Dissolution of nutrients from this ash can increase soil base cation concentrations and promote leaching (Grier 1975; Woodmansee & Wallach 1981; Boerner & Forman 1982). Such leaching is probably responsible for larger losses of base cations, Cl^-, SO_4^{2-} and metals from the rooted zone than were measured in this study.

Groundwater. The effect of infiltrating throughflow on groundwater chemistry was particularly apparent with the inorganic-N fractions. After burning, the concentration of NO_3^- in groundwater of the mid-slope wells in the experimental plot was higher by a factor of 57 (Figure 8) compared to that of the control plot wells, which remained relatively invariant. The lower groundwater [NO_3^-] compared to that of throughflow indicates that dilution and/or NO_3^- removal at depth occurred in both plots. Matson et al. (1987) showed that plant uptake, inorganic adsorption, and immobilization by microbes or soil organic matter are important mechanisms responsible for mineralized N retention in cut and burned plots at Turrialba, Costa Rica.

The subsequent increase of [NH_4^+] in the experimental plot after cutting was similar to that of NO_3^-, albeit more abrupt, and is followed by a decrease in concentration until the end of a period with uncharacteristically low rainfall. An abrupt increase in soil solution NH_4^+ followed by a rapid decline has been reported previously in the burning of temperate forests (St. John & Rundel 1976; Tomkins et al. 1991; Chorover et al. 1994). With the advent of larger rains toward the end of the dry season, a second pulse of NH_4^+ in the experimental plot wells occurred that was not seen in the control, probably a result of disturbance and infiltrating throughflow. The #7 well in the experimental plot had high [NH_4^+] throughout most of the study ($>50\ \mu M$), and may be why [NH_4^+] was much lower in throughflow than in groundwater in the experimental plot. It is also possible that either ammonification occurred at depth or our limited throughflow measurements underestimated the extent of NH_4^+ leaching in burned areas.

Baseline concentrations of NH_4^+ were more variable than those of NO_3^- in both plots and may be attributed to the process of well maturation (Figure 8). The convergence of $[NH_4^+]$ in the control and experimental plots toward the end of the 1989 wet season and prior to forest cutting suggests that the wells were stabilizing relative to NH_4^+ at this time. However, the time series of solute concentrations in groundwater show that stabilization was apparently not achieved for some solutes by the cutting phase of the experiment (Figure 9). In particular, concentrations of ANC, Ca^{2+} and Mg^{2+} show large differences in groundwater immediately prior to the cut. We believe the higher concentrations of some solutes in the wells of the experimental plot immediately prior to the cut was a result of fitting the top portion of the wells with a 0.5 m removable section. This procedure was done to allow the tops of the wells to be buried during the burning phase of the experiment, and apparently resulted in the penetration of some top soil into the vicinity of the well screens. However, the 3-month period in between cutting and burning allowed the wells to stabilize, and the effects of the burn were observed due to the larger flushing events of the wet season in early December 1989.

Despite possible disturbance effects from the above procedure, high groundwater [ANC] and its large between-plot differences observed before cutting the experimental plot were unexpected. Given the moderate acidity and low [DIC] of groundwater (Table 1), the contributions of HCO_3^- and CO_3^{2-} to groundwater ANC were trivial. Negligible carbonate alkalinity indicates that there were probably other dominant sources of groundwater ANC. Since the pK_as of most ions are influential at a pH <5.0, ANC may have resulted from the presence of organic acids or other DOC.

Alternatively, clay particles were commonly found in groundwater samples with large ANC. These may have been due to purging of the wells, and could have acted as a source of ANC because of their numerous exchange sites which can act as buffers. Colloidal clays appear to be the principal factor responsible for the between-plot differences and temporal changes observed in ANC in the groundwater of both plots. ANC resulting from the presence of colloidal clays was almost eliminated in the wells after about 6 months of post-cut sampling and continual flushing. These clays were more apparent in the wells of the cut plot than the control plot, and diminished throughout the study (pers. obs.).

Hydrogen ion concentrations in the groundwater of both plots gradually increased in parallel from the beginning to the pre-burn phase of the experiment (Figure 10). Post-burn $[H^+]$ decreased in the groundwater of the experimental plot and both plots plateaued for several months until February 1990, when concentrations rose to about 43 μeq L^{-1} and subsequently decreased to about 10 μeq L^{-1} by the end of the study. Thus pH in both plots varied from about 7.0 to 4.3 over the course of the experiment, with a period of several months after the burn when [H$^+$] diverged and pH was higher in the groundwater of the cut plot. We attribute the post-burn decrease in [H$^+$] of the experimental plot to the alkalinity generated from burning in mid-October. The pH of groundwater in both plots decreased to about 4.3 in March 1990, values that are well below the mean pH of 5.0 for rainfall in the wet season (Williams et al. 1997). Hydrogen ion and DOC concentrations parallel one another from February until March 1990 (Figure 10), indicating that changes in the pH of groundwater may be attributed to unmeasured organic acids.

DOC concentrations in both plots increased after the experimental plot was cut, but subsequently decreased in the control plot after the beginning of the wet season in December (Figure 10). In the experimental plot, [DOC] and [NO_3^-] are well correlated throughout the sampling period, which indicates that the increasing [DOC] measured in these wells was a result of infiltrating water from throughflow. Increases in [DOC] are commonly a result of elevated decomposition resulting from higher soil moisture and microbial activity (Meyer & Tate 1983; Tate & Meyer 1983). However, groundwater [DOC] in the control plot decreased below that measured in throughflow in December, suggesting that the onset heavy rains increased DOC removal processes or diluted DOC in groundwater.

Nitrogen evolution from groundwater to the stream. The high NO_3^- pulse from throughflow in the experimental plot never appeared in the wells located in the forested buffer strip bordering the stream or in measurements of bank seepage. Moreover, the ratio of NO_3^- to NH_4^+ decreased in groundwater from the upper slope to the riparian and hyporheic zones of the stream (Figure 11). For example, NO_3^- in the forested catchment was higher by a factor of 5 in the mid-slope wells (#2, 3, 7 and 8) compared to the upslope wells (#4 & 5), and lower in the streambank wells (#1 and 6). Nitrate in streambank springs was similar in concentration to that of the mid-slope wells, which may be macropore flow by-passing deeper zones of denitrification in the riparian buffer (not shown). Nitrate was lower in bank seepage collected with the seepage meters compared to the streambank well (Figure 11), while NH_4^+ was higher by about a factor of 2. McClain et al. (1994) suggest that NO_3^- disappearance in the anaerobic riparian sediments of the Ducke Reserve north of Manaus is probably due to denitrification, whereas NH_4^+ concentrations may be explained by ammonification uncoupled from nitrification. Patterns of isotopic composition in upland, riparian and stream waters observed by Brandes et al. (1996) suggest also that groundwater dissolved inorganic N

(DIN) is removed in the riparian zone, and that DIN in stream water is primarily remineralized organic N.

It is also possible that the NO_3^- losses occurring along the hydrologic pathway of groundwater to the sediment-stream interface was a result of uptake by aggrading riparian vegetation (Peterjohn & Correll 1984). High growth rates of trees occurred in the riparian area as a result of increased light availability after deforestation on the opposite side of the stream (Williams, unpubl. data). Moreover, larger groundwater flow patterns driven by large-scale infiltration in the Calado basin could have caused significant dilution of observed throughflow in the small-scale experimental plot. Groundwater probably moved both laterally in relation to the well transects and toward the lake, although the fraction of the contributing area influenced by the experimental treatment could not be determined.

In transition from the streambank sediments (seepage meters) to the stream, NO_3^- was higher by about 6 μM, whereas NH_4^+ was lower by about 6 μM (Figure 11). This speciation shift may be attributed to the higher redox potentials characteristic of a riparian-stream interface compared to anoxic riparian sediments. Ammonium entering the aerobic hyporheic zone is commonly oxidized to NO_3^- by nitrifiers (Blackburn 1983), and the commensurate $[NH_4^+]$ decrease associated with an increase in $[NO_3^-]$ suggests that this is the predominant microbial process. This mechanism probably acts in concert with uptake by aggrading vegetation.

Comparison with other studies. Large increases in the nutrient concentrations of soil leachates were observed by Uhl et al. (1982) and Uhl & Jordan (1984) after cutting and burning small plots of *caatinga* and *terra firme* rain forest, respectively, in San Carlos, Venezuela. Similarly, Parker (1985) observed increases in the concentrations of solutes in soil solutions in cleared, unburned rainforest areas of Volcan Barba, Costa Rica. Soils of one San Carlos study were infertile Spodosols (Uhl et al. 1982), while those of the Uhl & Jordan (1984) and Lake Calado study sites were Oxisols. Soils of the Volcan Barba site were identified by Parker as Ultisols. Annual rainfall was higher at San Carlos and Volcan Barba than at Lake Calado (3500 and 4000 versus 2800 mm, respectively).

Uhl & Jordan (1984) observed abrupt increases in NO_3^-, K^+, Ca^{2+} and Mg^{2+} after cutting and burning that subsided to pre-burn concentrations after about 2 years. The divalent cations at San Carlos showed considerable variability in the control plot leachates in comparison to the other solutes, and most peak concentrations occurred in the first year after the cut, except for Mg^{2+}, which peaked in the second year. Although we do not have a second year of data, trends indicate that the concentrations of most solutes were approaching control levels by the end of our study.

Uhl et al. (1982) observed higher peak concentrations of Ca^{2+}, Mg^{2+}, K^{+}, and much lower concentrations of NO_3^- compared to the studies above. Phosphate and NH_4^+ were much higher than concentrations observed in our study, and were not measured by Uhl & Jordan (1984). Large increases in soil solutions were observed about four months after clearcutting plots at Volcan Barba (Parker 1985), and maximum concentrations occurred after about 6 to 8 months and lasted for over one year.

Although the effects of burning forests on soil nutrient status (Smith 1970) and solute fluxes in stream water (Schindler et al. 1980; Feller & Kimmins 1984; Bayley et al. 1992a, b) are known, that we are aware of, no studies have documented the effects of cutting and burning mature, temperate forests on soil solution or groundwater chemistry. Boerner & Forman (1982) observed elevated concentrations of Ca^{2+}, Mg^{2+} and K^{+} in soil solutions after wildfire and prescribed burning, with the largest losses occurring after wildfire, but other solutes were not measured. Using porous-cup lysimeters, Chorover et al. (1994) observed increases in the concentrations of SO_4^{2-}, NO_3^- and Cl^- in soil solutions for up to 3 years after prescribed burning.

Comparison of these data with our study is not intended to exemplify the differences between tropical and temperate forest ecosystems, especially since the disturbances and sample sizes are different; rather these are the only studies for which this kind of information is available. However, the overall results of these studies illustrate some general similarities between tropical and temperate catchments. Solute concentrations increase rapidly after fire and return to background levels after several years, albeit with much variability among solutes. A combination of high rates of production and high element storage in successional vegetation (Vitousek & Reiners 1975) together with a decline in easily decomposable substrates is probably responsible for a reduction in leaching losses. Clearly, we can generalize about expected solute losses following disturbances involving fire, but predicting the magnitude of solute mobilization and its effect on soil fertility still depends on site specific characteristics of catchments, such as soil type, the extent of ground disturbance, and severity of the fire (Boerner & Forman 1982), all which cannot be determined without empirical data. Although the magnitude of the effects observed by Uhl & Jordan (1984) and our study are loosely transferable to other moist tropical catchments with Oxisols, studies using similar treatments and methodologies are required before a rigorous assessment of the different responses of tropical catchments to cutting and burning can be achieved. Moreover, because slash-and-burn disturbances in mature, temperate forests are uncommon, with most cutting followed by the removal of forest biomass for commercial purposes before slash is burned (Feller & Kimmins 1984), it is unlikely that a variety of suitable comparisons of soil water and groundwater from temperate-zone catchments will become available soon.

Acknowledgements

We thank the Instituto Nacional de Pesquisas da Amazônia (INPA) for logistical support in Brazil. Financial support was received from the National Geographic Society (4130-89) and the US National Science Foundation (BSR 87-06643). Data analysis was supported also by NASA grant NAGW-2602. We thank M. Moline for assistance in the field, and F. Setaro for soil analyses. Editorial contributions were made by K. Lajtha, H. Elsenbeer, and two anonymous reviewers. This manuscript was improved by the valuable suggestions of P. Matson.

References

Bayley SE & Schindler DW (1991) The role of fire in determining stream water chemistry in northern coniferous forests. In: Mooney HA, Medina E, Schindler DW, Schulze ED & Walker FH (Eds) Ecosystem Experiments, SCOPE 45 (pp 141–165). John Wiley & Sons, New York, 268 pp

Bayley SE, Schindler DW, Keaty KG, Parker PB & Stainton MP (1992a) Effects of multiple fires on nutrient yields from streams draining boreal forest and fen watersheds: Nitrogen and phosphorus. Can. J. Fish. Aquat. Sci. 49: 584–596

Bayley SE, Schindler DW, Parker BR, Stainton MP & Beaty KG (1992b) Effects of forest fire and drought on acidity of a base-poor boreal forest stream: Similarities between climatic warming and acidic precipitation. Biogeochemistry 17: 191–204

Blackburn TH (1983) The microbial nitrogen cycle. In: Krumbein W (Ed) Microbial Geochemistry (pp 63–89). Blackwell Scientific Publ., Oxford, 330 pp

Boerner RJ & Forman RT(1982) Hydrologic and mineral budgets of New Jersey Pine Barrens upland forests following two intensities of fire. Can. J. For. Res. 12: 503–510

Booth W (1989) Monitoring the fate of the forests from space. Science 243: 1428–1429

Brandes JA, McClain ME & Pimentel TP (1996) ^{15}N evidence for the origin and cycling of inorganic nitrogen in a small Amazonian catchment. Biogeochemistry (in press)

Chorover J, Vitousek PM, Everson DA, Esperanza AM & Turner D. (1994) Solution chemistry profiles of mixed-conifer forests before and after fire. Biogeochemistry 26: 115–144

Fearnside PM (1989) A prescription for slowing deforestation in Amazonia. Environ. Mag. 31: 17–41

Fearnside PM (1990) Deforestation in the Brazilian Amazon. In: Woodwell GM (Ed) The Earth in Transition: Patterns and Processes of Biotic Impoverishment (pp 211–238). Cambridge Univ. Press 530 pp

Feller MC & Kimmins JP (1984) Effects of clearcutting and slash burning on stream water chemistry and watershed nutrient budgets in southwestern British Columbia. Wat. Resour. Res. 20: 29–40

Gosz JR (1981) Nitrogen cycling in coniferous ecosystems. In: Clark RE & Rosswal T (Eds) Terrestrial Nitrogen Cycles (pp 405–426). Ecological Bulletins-EFR 33, 714 pp

Grier CC (1975) Wildfire effects on nutrient distribution and leaching in a coniferous ecosystem. Can. J. For. Res. 5: 599–607

Hamilton LS & King PN (1983) Tropical Forested Watersheds Hydrologic and Soils Response to Major Uses or Conversions. West-view Press Inc., Boulder, CO, 168 pp

Hart SE & Firestone MK (1991) Forest floor – mineral soil interactions in the internal nitrogen cycle of an old-growth forest. Biogeochemistry 12: 103–127

Hodnett MG, Pimentel da Silva L, da Rocha HR & Cruz Senna R (1995) Seasonal soil water storage changes beneath central Amazonian rainforest and pasture. J. Hydro. 170: 233–254

Houghton RA (1990) The future role of tropical forests in affecting the carbon dioxide concentration of the atmosphere. Ambio 19: 204–209

Johnson DW, Cole DW, Gessel SP, Singer MJ & Midden RV (1977) Carbonic acid leaching in a tropical, temperate, subalpine and northern forest soil. Arct. & Alp. Res. 9: 329–343

Jordan CF (1987) Amazonian Rain Forests: Ecosystem Disturbance and Recovery. Springer-Verlag, 133 pp

Lee DR & Cherry JA (1979) A field exercise on groundwater flow using seepage meters and mini-peizometers. J. Geol. Educ. 27: 6–10

Leopoldo PR, Franken W & Salati E (1982) Balanco hidrico de pequena bacia hidrografica em floresta Amazonic de terra firme. Acta Amaz. 12: 333–337

Lesack LFW (1993a) Export of nutrients and major ionic solutes from a rain forest catchment in the central Amazon Basin. Wat. Resour. Res. 29: 743–758

Lesack LF (1993b) Water balance and hydrologic characteristics of a rain forest catchment in the central Amazon basin. Wat. Resour. Res. 29: 759–773

Lesack LFW & Melack JM (1991) The deposition, composition, and potential sources of major ionic solutes in rain of the central Amazon basin, Wat. Resour. Res. 27: 2953–2978

Likens GE, Bormann FH, Johnson NM, Fisher DW & Pierce RS (1970) Effects of forest cutting and herbicide treatments on nutrient budgets in the Hubbard Brook watershed-ecosystem. Ecol. Monogr. 40: 23–47

Lobert JM, Scharffe DH, Hao WM, Kuhlbusch TA, Seuwen R, Warneck P & Crutzen PJ (1991) Biomass burning emissions: Nitrogen and carbon containing compounds. In: Levine JS (Ed) Global Biomass Burning (pp 289–304). MIT Press, Cambridge, 569 pp

Matson PA & Vitousek PM (1981) Nitrification potentials following clearcutting in the Hoosier National Forest, Indiana. Forest Science 27: 781–791

Matson PA, Vitousek PM, Ewel JJ, Mazzario MJ & Robertson GP (1987) Nitrogen transformations following tropical forest felling and burning on a volcanic soil. Ecology 68: 491–502

McClain ME, Richey JE & Pimentel TP (1994) Groundwater nitrogen dynamics at the terrestrial-lotic interface of a small catchment in the central Amazon basin. Biogeochemistry 27: 113–127

Meyer JL & Tate CM (1983) The effects of watershed disturbance on dissolved organic carbon dynamics of a stream. Ecology 64: 33–44

Muller-Dombios D (1990) Impoverishment in Pacific Islands. In: Woodwell GM (Ed) The Earth in Transition: Patterns and Processes of Biotic Impoverishment (199–210). Cambridge Univ. Press, 530 pp

Nortcliff S & Thornes JB (1978) Water and cation movement in a tropical rainforest environment. Acta Amaz. 8: 245–258

Nortcliff S & Thornes JB (1981) Seasonal variations in the hydrology of a small forested catchment near Manaus, Amazonas, and the implications for its management. In: Lal R & Russel EW (Eds) Agricultural Hydrology (pp 37–57). John Wiley & Sons, Chichester, UK, 149 pp

Nortcliff S, Thornes JB & Waylen MJ (1979) Tropical forest systems: A hydrological approach. Acta Amaz. 6: 557–558

Page AL, Miller RH & Keeney DR (1982) Methods of Soil Analysis. American Society of Agronomy, Soil Science Society of America, 1159 pp

Parfitt RL (1978) Anion adsorption by soils and soil materials. Adv. Agro. 30: 1–50

Parker GG (1985) The effect of disturbance on water and solute budgets of hillslope tropical rainforest in northeastern Costa Rica. Ph.D. dissertation, Univ. Georgia, 161 pp

Peterjohn WT & Correll DL (1984) Nutrient dynamics in an agricultural watershed: Observations on the role of a riparian forest. Ecology 65: 1466–1475

Raison RJ, Keith J & Khanna P (1990) Effects of fire on the nutrient-supplying capacity of forest soils. In: Dyck W & Mees M (Eds) Impact of Intensive Harvesting on Forest Site Productivity. Forest Research Institute Bulletin No 159, Rotorua, New Zealand

Salati E & Vose PB (1984) Amazon basin: A system in equilibrium. Science 225: 129–138

Schindler DW, Newbury RW, Beaty KG, Prokopowich J, Ruszczynski T & Dalton JA (1980) Effects of a windstorm and forest fire on chemical losses from forested watersheds and on the quality of receiving streams. Can. J. Fish. Aq. Sci. 50: 17–29

Schock P & Binkley D (1986) Prescribed burning increased nitrogen availability in a mature loblolly pine stand. For. Ecol. Man. 14: 13–22

Shepard JP, Mitchell MJ, Scott TJ & Driscoll CT (1990) Soil solution chemistry of an Adirondack spodosol: lysimetry and N dynamics. Can. J. For. Res. 20: 818–824

Sollins P, Grier CC, McCorison FM, Cromack K, Fogel R & Fredriksen RL (1980). The internal element cycles of an old-growth Douglas-fir ecosystem in western Oregon. Ecol. Monog. 50: 261–285

Sposito G (1989) The Chemistry of Soils. Oxford Univ. Press, New York, 277 pp

St. John TV & Rundel PW (1976) The role of fire as a mineralizing agent in a Sierran coniferous forest. Oecologia 25: 35–45

Stark NM & Jordan CF (1978) Nutrient retention by the root mat of an Amazonian rain forest. Ecology 59: 434–437

Staver K, Magette W & Brinsfield R (1987) Tillage effects on nutrient and sediment field losses. ASAE paper 87-2086, American Society of Agricultural Engineers, St. Joseph, MI

Swank WT, Swift Jr LW & Douglass JE (1988) Streamflow changes associated with forest cutting, species conversions, and natural disturbances. In: Swank WT & Crossley DA (Eds) Forest Hydrology and Ecology at Coweeta (pp 297–312). Springer-Verlag, 469 pp

Tardin AT & da Cunha RP (1990) Evaluation of deforestation in the legal Amazonia using Landsat-TM images. INPE-5015-RPE/609

Tate CM & Meyer JL (1983) The influence of hydrologic conditions and successional state on dissolved organic carbon export from forested watersheds. Ecology 64: 25–32

Tiedemann AR, Conrad CE, Dietrich JH, Hornbeck JW, Megehan WF, Viereck LA & Wade DD (1978) Effects of Fire on Water: A State of Knowledge Review. USDA Forest Service General Technical Report WO-10

Tomkins IB, Kellas JD, Tolhurst KG & Oswin DA (1991) Effects of fire on soil chemistry in a Eucalypt forest. Aust. J. Soil Res. 29: 25–47

Ugolini RC, Minden R, Dawson H & Zachara F (1977) An example of soil processes in the *Abies amabilis* zone of the Central Cascades, WA. Soil Sci. 124: 291–302

Uhl C & Jordan CF (1984) Succession and nutrient dynamics following forest cutting and burning in Amazonia. Ecology 65: 1476–1490

Uhl C, Jordan C, Clark K, Clark H & Herrera R (1982) Ecosystem recovery in Amazon caatina forest after cutting, cutting and burning, and bulldozer clearing treatments. Oikos 38: 313–320

Vegas-Vilarrúbia T, Maas M, Valentí R, Eliás V, Ovalle ARC, Lopez D, Schneider G, Depetris PJ & Douglas I (1994) Small catchment studies in tropical zone. In: Moldan B & Cerny J (Eds) Biogeochemistry of Small Catchments, SCOPE 51 (pp 343–360). John Wiley & Sons, 419 pp

Vitousek PM (1980) Nitrogen losses from disturbed ecosystems – ecological considerations. In: Rosswall T (Ed) Nitrogen Cycling in West African Ecosystems (pp 39–53). Royal Swedish Academy of Sciences, 450 pp

Vitousek PM & Reiners WA (1975) Ecosystem succession and nutrient retention: A hypothesis. Bioscience 25: 376–381

Vitousek PM & Matson PA (1985) Disturbance, nitrogen, availability, and nitrogen losses in an intensively managed loblolly pine plantation. Ecology 66: 1360–1376

Vitousek PM, Gosz JR, Grier CC, Mellilo JM, Reiners WA & Todd RC (1979) Nitrate losses from disturbed ecosystems. Science 204: 469–474

Wallace JB (1988) Aquatic invertebrate research. In: Swank WT & Crossley Jr DA (Eds) Forest Hydrology and Ecology at Coweeta (pp 257–268). Springer-Verlag, 469 pp

Williams MR (1993) The effects of deforestation on the water chemistry of a small watershed in central Amazonas. MS thesis, Univ. of Maryland, 254 pp

Williams MR & Melack JM (1997) Solute export from forested and partially deforested catchments in the central Amazon. Biogeochemistry 38: 67–102

Williams MR, Fisher TR & Melack JM (1997) The composition and deposition of rain in the central Amazon, Brazil. Atm. Env. 31: 207–217

Woodmansee RG & Wallack LS (1981) Effects of fire regimes on biogeochemical cycles. In: Clark FE & Rosswall T (Eds) Terrestrial Nitrogen Cycles (pp 649–669). Ecological Bulletins, Stockholm

Woodwell GM, Houghton RA, Stone TA & Park AB (1986) Changes in the area of forests in Rondônia, Amazon Basin, measured by satellite imagery. In: Trabalka J & Reichle K (Eds) The Changing Carbon Cycle: A Global Analysis. Springer-Verlag, New York, 672 pp

Wright RF (1976) The impact of forest fire on the nutrient influxes to small lakes in northeastern Minnesota. Ecology 57: 649–663