Biogeochemistry **38:** 67–102, 1997.

Solute export from forested and partially deforested catchments in the central Amazon

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Key words: Amazon, catchments, deforestation, slash-and-burn agriculture, solutes, water balance

Abstract. The hydrochemical responses to slash-and-burn agriculture in a small rainforest catchment of the central Amazon were investigated for one year. Disturbances in the partially deforested catchment began in 1987, and during the study a 2-ha plot was cut (July 1989) and burned (October 1989) in preparation for the cultivation of manioc; the partially deforested catchment was approximately 80% deforested at the time of this study. Solute fluxes exported by base flow were estimated from solute concentrations of stream water measured at least once per week. Solute fluxes for storm flow were estimated by measuring streamwater concentrations during two storms. Baseflow runoff represented about 94% of the water outflow from the study basin and was the dominant pathway of solute export. Total rainfall during the study period was 2754 mm of which 2080 mm was exported from the partially deforested catchment as stream runoff. The ratio of surface runoff to annual rainfall for a similar study conducted in the same catchment while completely forested in 1984 was lower than after the catchment was 80% deforested in 1990 (0.57 versus 0.76), while evapotranspiration (ET) was lower by about a factor of two in 1990 compared to 1984. Particulate removal from the partially deforested catchment was 151 kg ha^{-1} yr^{-1}. Nutrient losses from the partially deforested catchment were higher than those measured when the catchment was undisturbed in 1984 by factors of 1.4, 1.8, and 2.1 for total inorganic nitrogen (TIN), total dissolved nitrogen (TDN), and total nitrogen (TN); and by factors of 4.0, 6.6, and 7.9 for soluble reactive phosphate (PO_4^{3-}), total dissolved phosphorus (TDP), and total phosphorus (TP), respectively. These data show that deforestation and colonization in upland catchments of the central Amazon alter the hydrochemical balance of streams by decreasing ET, thereby increasing discharge and solute export.

Introduction

Slash-and-burn agriculture is a pervasive method of deforestation in the Amazon and is increasing (Booth 1989). Tardin & da Cunha (1990) estimate that about 300,000 km^2 of Amazon forest was cut from 1960 to 1989, and such large-scale deforestation has implications concerning the water balance (Salati & Vose 1984), carbon cycle (Houghton 1990), and extinction of species in the Amazon basin (Fearnside 1990; Muller-Dombois 1990). There are also important localized effects on catchments attributed to deforestation. These include reduced transpiration, and increased runoff generation and water yields (Dubreuil 1985; Swank et al. 1988; Bruijnzeel 1990), the development of erosional processes that increase soil losses, and enhanced leaching of soluble materials (Likens et al. 1970; Vitousek et al. 1979; Vitousek 1980; Bruijnzeel 1990; Williams et al. 1997a). Downstream receiving waters are influenced by the increased inputs of water and solutes, and N and P from cut areas are of particular concern because of their role in aquatic eutrophication. Higher nutrient inputs to the receiving waters of temperate ecosystems commonly cause increased algal abundance and changes in species composition (Bormann & Likens 1970; Wallace 1988). Presently, most information concerning the effects of deforestation comes from the temperate zone, and is not necessarily applicable to the tropics due to site specificity (Janzen 1973; di Castri & Hadley 1979; Vitousek et al. 1979; Vitousek & Sanford 1986). Research concerning the effects of deforestation in tropical areas remains sparse (Hamilton & King 1983; Uhl & Jordan 1984; Jordan 1987; Uhl et al. 1989; Bruijnzeel 1990).

Slash-and-burn agriculture requires a ratio of cultivation to fallow periods small enough to allow adequate regeneration of the soils and vegetation (Jordan 1987). However, increasing population density, and concomitant political, economic, and social pressures within Brazil have created a more intensive form of slash-and-burn agriculture characterized by shorter fallow periods and additional crop plantings (Hecht 1982). At the time of our study, the Brazilian government attempted to mitigate intensive slash-and-burn agriculture through the use of educational seminars provided by the Instituto Nacional de Colonização e Reforma Agraria (INCRA). Riparian buffer strips were required, and protocols stipulated that three crop rotations of manioc over two years should be followed by a minimum fallow period of 5 years, but enforcement was rare. Consequently, some of the study site basin lacked riparian buffers, and instances of terrestrial solute losses and erosion were observed (Williams et al. 1997a).

The purpose of our study was to measure the effects of slash-and-burn agricultural practices and colonization on the hydrochemistry of streams in small rainforest catchments in the central Amazon. The study site was a first-order stream in an upland catchment largely deforested as a result of slash-and-burn agriculture used for the purpose of cultivating manioc. Comparisons of stream water are made with similar measurements from an adjacent undisturbed catchment stream and from a previous study by Lesack (1993a; b) describing the partially deforested catchment in 1984, prior to cutting. Our results provide evidence that deforestation and colonization in the central

Amazon alter the solute chemistry and water balance of streams draining disturbed areas.

Study site

Lake Calado is located in the central Amazon basin (3°15′ S, 60°34′ W) on the north bank of the Solimões River (Amazon main stem) about 80 km W of Manaus (pop. ca. 1,000,000 in 1990 (IBGE 1992); Figure 1). It is a floodplain lake connected to the Solimões River year round and undergoes changes in depth from 1 to 12 m over the course of the annual hydrological cycle. The study site is an upland catchment (23.4 ha) with a perennial first-order stream called the Braço do Mota, and was completely forested as of 1984. An adjacent forested catchment (18 ha) with a stream called the Igarapé de Mota (Moto Brook) was used as a control. In 1987, the Brazilian government promoted settlement of the eastern lake basin by constructing a road connecting Lake Calado to the main road running between Manaus and Manacapuru (Figure 1). Increased accessibility provided by the road allowed settlers to move into the area, and these settlers relied on slash-and-burn agriculture to grow manioc. Eighty percent of the Braço do Mota catchment had been converted to agricultural plots at the time of our study. Cutting and burning of the Braço do Mota catchment (henceforth referred to as the partially deforested catchment) during our study was restricted to a 2-ha plot that was cut in mid-July 1989. Trees were felled by ax and ground disturbances were minimal. The cut forest was burned in mid-October 1989, and some of the unburned wood was removed from the plot before planting manioc in early November 1989. Detailed descriptions of the study site are given in Williams et al. (1997a; b) and Lesack (1993a; b).

Methods

Measurements of rainfall. Rainfall was measured using a combination of a tipping bucket rain recorder and manual gauges (Williams et al. 1997a). A total of 4 manual gauges were placed in the partially deforested catchment (Figure 2). Two gauges were read on an event basis, while the others were read at least twice per week. Manioc around these gauges was pruned regularly to eliminate possible interferences, and the methods and analyses used for rainfall in this study are summarized in Table 1.

Measurements of stream discharge. Stream discharge was measured at the 1990 weir site with a sharp crested 120° V-notch weir and stilling well

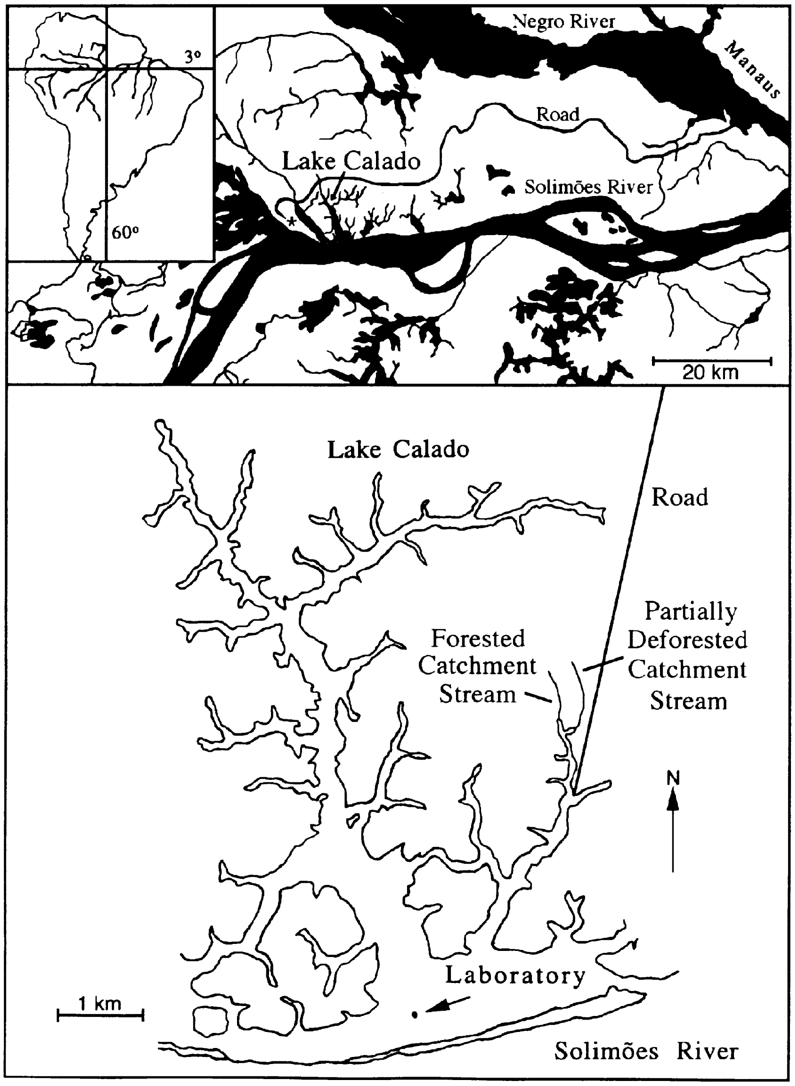

Figure 1. Map of South America, the central Amazon and the Lake Calado basin with designated partially deforested (experimental) and forested (control) catchment streams. The city of Manacapuru is designated by an asterisk.

equipped with a water-level recorder. The water-level recorder had a strip chart for continuous measurements of discharge, and a plastic pipe 1.3 cm in diameter extended from the stilling well to a ponding basin to record water head without surface drawdown effects at the crest of the weir (Stevens 1987). The weir was calibrated over the entire range of baseflow runoff (5 to 15 L

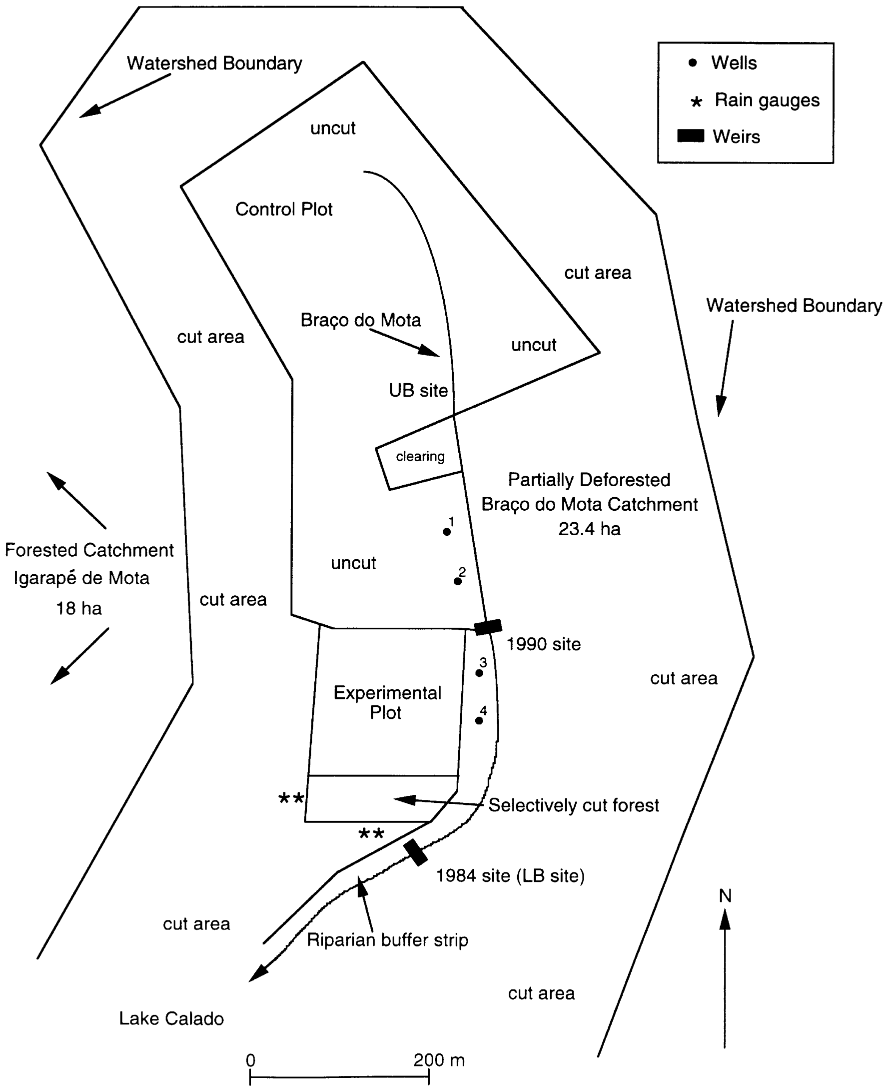

Figure 2. The partially deforested catchment indicating equipment locations and the experimental design of the plot experiment (Williams et al. 1997a). The Igarapé de Mota, or Mota Brook (MB) catchment is directly to the west of the partially deforested catchment.

s^{-1}) against independent measurements recorded with a large plastic bucket at low flows and current velocity at higher flows. A manometer was used to determine water head in the ponding basin, and these data were used in conjunction with a commercial hydraulic table to calculate discharge (Stevens 1987). Given the design of the weir, the large size of the ponding basin, and

Table 1. Chemical analyses performed on water samples collected at Lake Calado, Brazil, during 1989–1990. Asterisks indicate that prewashed Gelman A/E glass fiber filters were used. Filtered and unfiltered trace metal samples were stored with Ultrex HNO_3. All filtered trace metal samples were passed through 0.1 μm Nuclepore filters. TDN and TN represent the total dissolved and total fractions of nitrogen; TDP and TP of phosphorus. "Y" and "N" indicate yes and no, respectively.

Constituent	Filtered	Analytical technique	Reference
pH	N	Orion pH meter with Ross glass combination electrode	Galloway et al. 1979
Conductivity	N	Yokogawa & YSI conductivity meters	Instrument Manual
Acid neutralizing capacity (ANC)	N	Gran titration using micro-meter burette and Orion pH meter	Gran 1950; 1952
Ammonium (NH_4^+)	Y*	Indophenol blue	Strickland & Parsons 1972
Phosphate (PO_4^{3-})	Y*	Molybdenum blue	Strickland & Parsons 1972
Nitrate (NO_3^-)	Y*	Cadmium reduction column	Wood et al. 1967
DOC	Y	Gas chromatography (Carle)	Stainton 1973
DIC	N	Gas chromatography (Carle)	Stainton 1973
TDN/TDP	Y*	Combined persulfate digestion	Valderrama 1981
TN/TP	N	Combined persulfate digestion	Valderrama 1981
Silicate	Y*	Silico-molybdate	Strickland & Parsons 1972
Anions (Cl^-, SO_4^{2-})	Y*	Ion chromatography (Dionex)	Instrument Manual
Cations (Na^+, K^+, Ca^{2+}, Mg^{2+})	Y*	Atomic absorption spectrophotometry (Varian)	Instrument Manual
Trace metals (Al, Fe, Mn)	Y & N	Atomic absorption with graphite furnace (Varian)	Instrument Manual

the low flows measured, the larger flow rates that occurred during storm events (15 to 47 L s^{-1}) were considered to be a reasonable extrapolation of baseflow runoff calculated using a commercial hydraulic chart.

Extrapolation of discharge measured at the upstream weir site (1990) to a lower reach of the stream where a weir was located in 1984 (Figure 2) was done to compare discharge for the two studies. An exponential relationship was calculated using measurements of instantaneous discharge at the 1990 weir site to extrapolate to the 1984 weir site. Instantaneous discharge was measured at the downstream location (1984 weir site) using salt dilution, and stream velocity and cross-sectional area (Buchanan & Summers 1969) over the entire range of flow encountered at the 1990 weir site. The ratio of the dissolved solids (mg L^{-1}) in streamwater over the interval of the salt dilution to the dissolved solids in the salt slug was calculated, and recovery of the salt slug was 100 $\pm$ 8% of the total for all samples used in the exponential rating curve for the 1984 weir site. A total of seven duplicated salt-dilution measurements were taken over the full range of baseflow runoff at the downstream weir site (1984), and replicates were always ±5%. Two of these measurements were compared with discharge calculations using cross-sectional area and stream velocity, and these measurements were within 12% of each other. Storm flow at the highest rates of discharge encountered was calculated using cross-sectional area and stream velocity estimates.

Measurement of stream flow and estimation of runoff. Estimates of baseflow and stormflow runoff were determined by the hydrograph separation technique of Hewlett & Hibbert (1967). Daily mean baseflow runoff was calculated using hourly readings from the strip charts. Storm flow was recorded at intervals of 20 minutes on either side of peak flow, and the results were combined with base flow to give a daily discharge estimate. Using 5 minute intervals on 2 large storms did not change the amount of storm flow calculated by more than ±5% because the peak of the stormflow hydrograph was incorporated into all estimates. Stormflow runoff measurements calculated from the strip chart recordings at the 1990 weir site were extrapolated to the 1984 weir site by calculating the stormflow yields from two storms at the 1984 weir site. These yields were used to formulate an equation which was applied against yields for the entire suite of storm events recorded at the 1990 weir site. The two storms were 9 and 58 mm and approximate the lower and upper ends of the 1990 storm distribution (range = 2 to 72 mm). Discharge for the 9 mm storm was calculated at the 1984 weir site using salt dilution measurements, whereas discharge of the 58 mm storm was calculated using duplicated cross-sectional area and stream velocity data (n = 3). Estimates for both storms included a measurement taken at peak discharge, which was determined using a temporary stream gauge. Since a vertical-velocity curve of the stream was not measured, the larger storm's stream velocity was adjusted for frictional effects of the streambank and bottom by taking 80% of the original discharge estimate. Buchanan & Summers (1969) suggest using a 15% reduction for frictional effects when mid-strata velocity data are available. It is assumed herein that an additional 5% reduction would account for the lower velocity surface measurements.

Baseflow discharge for the adjacent forested catchment stream of Mota Brook (henceforth referred to as MB) was calculated with the method used above for extrapolating to the downstream reach of the partially deforested catchment. Estimates of baseflow discharge from the mouth of the forested catchment were calculated throughout the study (n = 11) using duplicated stream velocity and cross-sectional area measurements. An exponential curve was calculated using the instantaneous measurements of discharge at MB against simultaneous measurements of base flow from the partially deforested catchment (1990 weir site). The equation was then used to calculate daily baseflow discharge for the forested catchment by applying the rating curve against the daily discharge from the 1990 weir site of the partially deforested catchment. We assume that stormflow discharge is about 6% of baseflow discharge in the forested catchment using the estimate of stormflow discharge calculated by Lesack (1993b) for the partially deforested catchment when it was completely forested in 1984.

Collection and chemical analysis of water samples. Stratified sampling of streamwater was performed to estimate the annual flux of solutes exported from the partially deforested catchment. The temporal variance of streamwater chemistry was assessed by taking samples on a weekly basis during baseflow conditions (n = 59). A series of samples was taken over the rising and falling hydrograph limbs during two storm events to evaluate streamwater solute dynamics and possible hysteresis (n = 19).

Particulate samples used for the analysis of C, N and P were filtered using ashed and weighed 47 mm Gelman A/E glass fiber filters with a nominal pore size of about 1 μm. After filtration, filters were oven dried and stored in a desiccator. Particulate export was measured by weighing the dried filters. Particulate C and N were determined in a modified Control Data Equipment Corp. CHN analyzer (model 2408) and particulate P was measured by high temperature combustion (Andersen 1976). Other methods used for analyzing streamwater are those summarized in Table 1.

Sample site locations. Stream samples were collected from a variety of locations. The stream of the partially deforested catchment was divided into upstream (forested) and downstream (deforested) sampling stations (Figure 2). Downstream samples were collected at the location of the 1984 weir site, which is henceforth referred to as the Lower Braço (LB) site. Upstream samples were collected at what will henceforth be referred to as the Upper Braço (UB) site. The adjacent forested catchment (MB) was sampled biweekly.

Results

Stream discharge and hydrograph characteristics of the partially deforested catchment. The time series of rainfall and runoff for the partially deforested catchment had monthly variations due to the seasonal pattern of rainfall (Figure 3). Precipitation is generally highest from December through May (wet season), and this period is one of net recharge to groundwater. The decreasing limb of the runoff hydrograph occurred from June through November (dry season), and the peak of the annual runoff hydrograph lagged

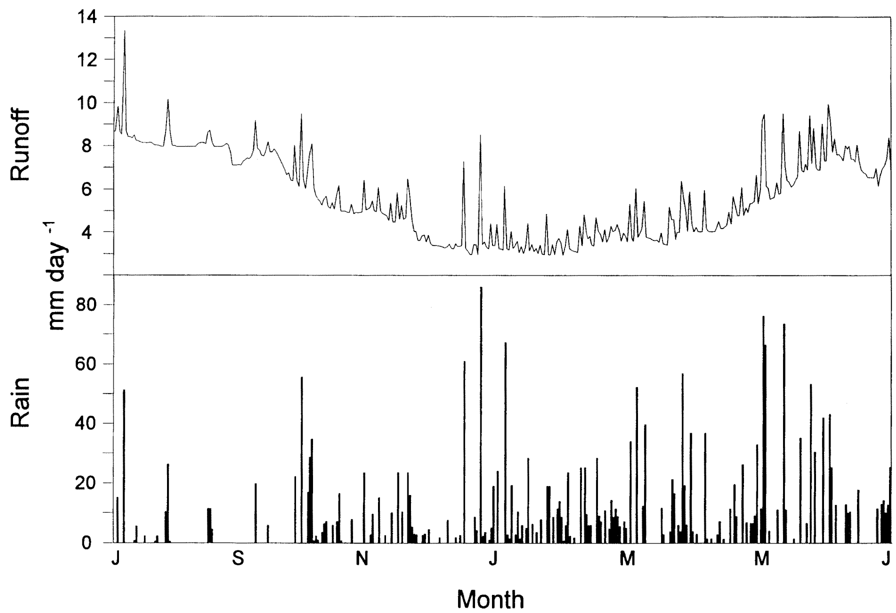

Figure 3. Annual distributions of rainfall and runoff, July 1989 to July 1990, in the partially deforested catchment.

behind that of rainfall by up to two months. For the period from July 1, 1989, to June 30, 1990, annual rainfall and streamwater runoff were 2754 and 2080 mm, respectively.

On an annual basis, stormflow runoff accounted for only 6% of the total discharge. Stormflow runoff ranged from 0.2 to 5.4 mm, but 73% of the storms that occurred produced less than 1 mm of storm flow. Of the total events recorded by the rain gauges, 61% yielded stormflow runoff hydrographs. Large storms generating more than 1 mm of stormflow runoff (27% of total storms) produced over 64% of total storm flow. The partially deforested catchment had an annual runoff coefficient of 0.76, and discharge was dominated by base flow (94%).

Baseflow and stormflow chemistry: Comparison of the Upper and Lower Braço sampling stations. A repeated-measure ANOVA was used to determine differences in baseflow solute concentrations between the UB and LB sampling stations. Concentrations of the N fractions measured at the UB were similar to those of the LB (Table 2), and TDN to TN ratios differed by only 9%. Concentrations of the P fractions at the UB were significantly lower than those of the LB by about 30% ($p < 0.05$). Most other ions were slightly lower in concentration at the UB compared to the LB. In contrast, $[H^+]$ at the UB

Table 2. VWM solute concentrations and sample sizes for streamwater at base flow for the three sampling stations in the study area. Values were determined from samples collected from July 1989 to July 1990. All units are μM, except ANC (μeq L^{-1}) and Mn (nM). Trace metals are separated into filtered (f) and unfiltered (u) sample sets. Baseflow particulate export is designated by "PE".

		Lower Braço		Upper Braço		Mota Brook	
Solute		VWM	n	Mean	n	Mean	n
H^+		11.1	50	25.5	37	19.1	27
NH_4^+		1.0	57	1.7	38	0.3	27
NO_3^-		11.7	58	11.9	26	8.2	27
TDN		19.7	52	19.4	37	11.6	26
TN		24.3	26	26.3	24	16.1	9
PO_4^{3-}		0.06	53	0.04	37	0.05	26
TDP		0.48	53	0.40	37	0.29	25
TP		0.61	32	0.52	24	0.31	9
Na^+		15.6	59	12.0	36	10.6	27
K^+		3.9	59	2.5	36	1.3	28
Ca^{2+}		4.0	50	3.2	29	2.8	27
Mg^{2+}		2.2	50	1.9	29	1.2	27
Cl^-		14.7	57	14.7	36	11.9	26
SO_4^{2-}		1.8	57	1.8	26	1.7	26
Si		80.4	43	70.1	26	89.9	27
pH		5.0	50	4.6	37	4.7	27
μS cm^{-1}		8.4	49	10.2	33	9.3	27
ANC		4.3	49	−5.6	33	−10.2	27
DOC		158	28	83	19	73	17
DIC		0.42	10	0.80	5	0.44	3
Al	f	1.5	24	2.2	20	1.9	20
	u	4.2	25	3.8	22	3.6	20
Fe	f	0.7	24	0.9	20	0.5	20
	u	3.0	25	2.8	22	1.5	20
Mn	f	55	24	36	20	36	20
	u	36	25	36	22	36	20
PE (kg ha^{-1} yr^{-1})		88.9		22.2		13.3	

was significantly higher than that of the LB by more than a factor of 2 ($p < 0.05$), and ranged from 6 to 174 μeq L^{-1}. Filtered trace metal samples were higher in [Al] and [Fe] at the UB, whereas [DOC] and [DIC] at the UB were half and more than twice those of the LB site, respectively. Mean

[DIC] measured during two longitudinal transects of the partially deforested catchment stream decreased from 1.1 at the UB to 0.4 μM at the LB sampling station.

Baseflow and stormflow chemistry of the Lower Braço sampling station. Solute concentrations of storm flow were up to 6 times higher than those of base flow (Figures 4 & 5), and stormflow export varied from 1 to 61% of the total annual export, not including subsurface runoff. Concentrations of N and P in storm flow were more than twice those in base flow, except for PO_4^{3-}. A large proportion of the TP export (Σ of PP and TDP) during base flow resulted from PP (Figure 4).

All the N fractions at the LB had positive relationships of baseflow discharge against solute concentration, except NH_4^+. There were no significant discharge-concentration relationships for any of the P fractions. Hydrogen ion had positive ($p < 0.05$), ANC and Mg^{2+} had negative ($p < 0.05$), and all other solutes had discharge-concentration relationships that were not significant.

Baseflow chemistry: Comparison of the partially deforested and forested catchments. A repeated-measure ANOVA, used to determine differences in solute concentration between catchment streams, showed that all of the N and P data (not including PO_4^{3-}) were statistically different ($p < 0.05$). Ammonium constituted only a small fraction of the total N removed annually from these catchments during base flow. The export of NH_4^+ was 3 and 2% for the partially deforested (LB site) and forested catchments (MB site), respectively. Nitrate accounted for 40% of the TN exported from the partially deforested catchment and 51% from the forested catchment. DON and PN fractions in MB were 27 and 28% of TN, respectively, which are lower and equal to those at the LB site (19 and 28% of TN, respectively). Phosphate export was 13% of TP at both sites. PP accounted for over 48% of the TP export estimate in the partially deforested catchment, compared to 7% in the forested catchment.

pH and [ANC] were significantly ($p < 0.05$) lower in MB compared to the LB site. Sulfate and Cl^- were similar between catchment streams, although there were significant differences ($p < 0.05$) between the base cations, especially K^+. Annual rainfall inputs accounted for only 15, 18, 36, and 24% of the Na^+, K^+, Ca^{2+} and Mg^{2+} exported from the partially deforested catchment, respectively, which are much lower than those of the forested catchment (39, 106 (i.e. net retention of K^+), 73 and 51%). Silicate concentrations were significantly lower ($p < 0.05$) at the LB site compared to those of MB (Table 2). Mean [DOC] in the LB was twice as high as in MB, whereas [DIC] in MB was slightly higher than that of the LB. Filtered trace metal concentrations had no significant differences between sites.

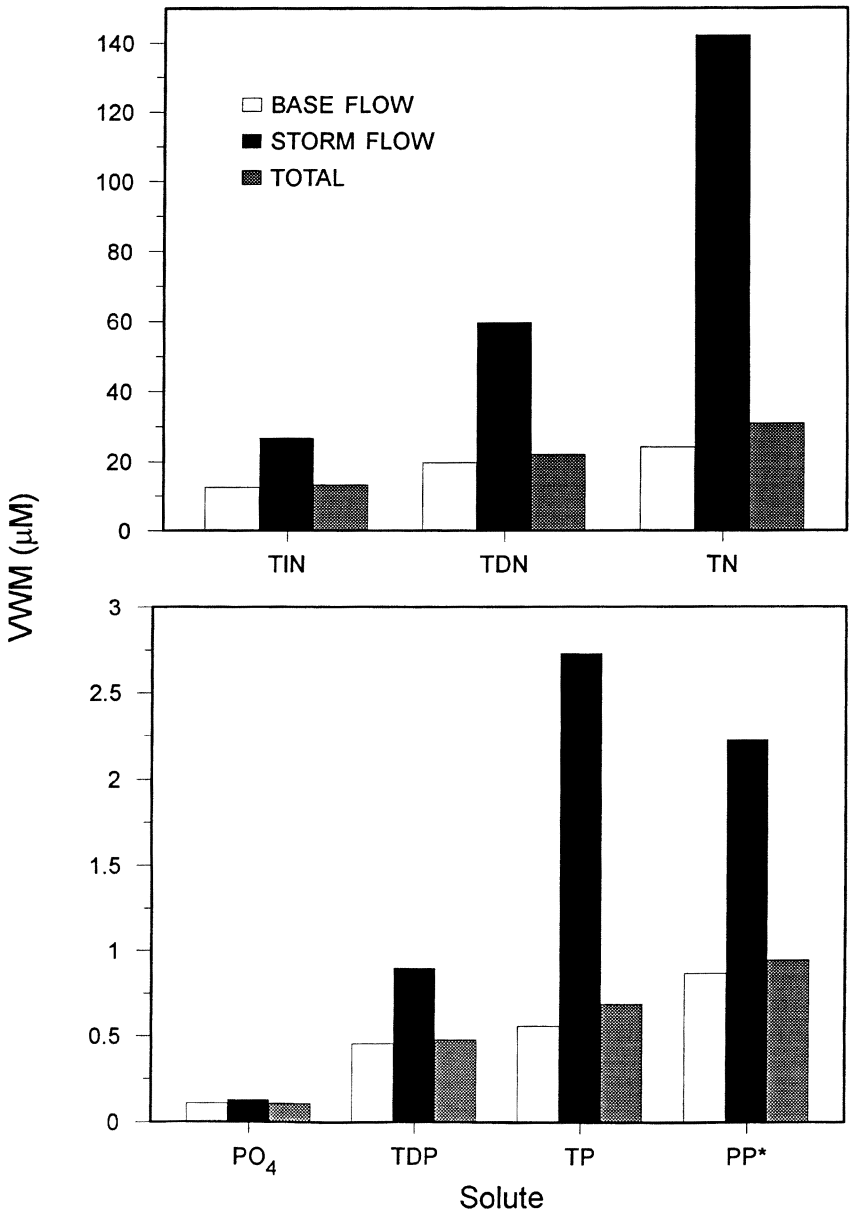

Figure 4. Volume-weighted mean concentrations of N and P fractions for base flow, storm flow and total discharge, July 1989 to July 1990. N and P fractions are: total inorganic nitrogen (TIN), total dissolved nitrogen (TDN), total nitrogen (TN), soluble reactive phosphate (PO_4^{3-}), total dissolved phosphorus (TDP), and total phosphorus (TP). The asterisk indicates that particulate phosphorus from the Andersen ignition technique was used.

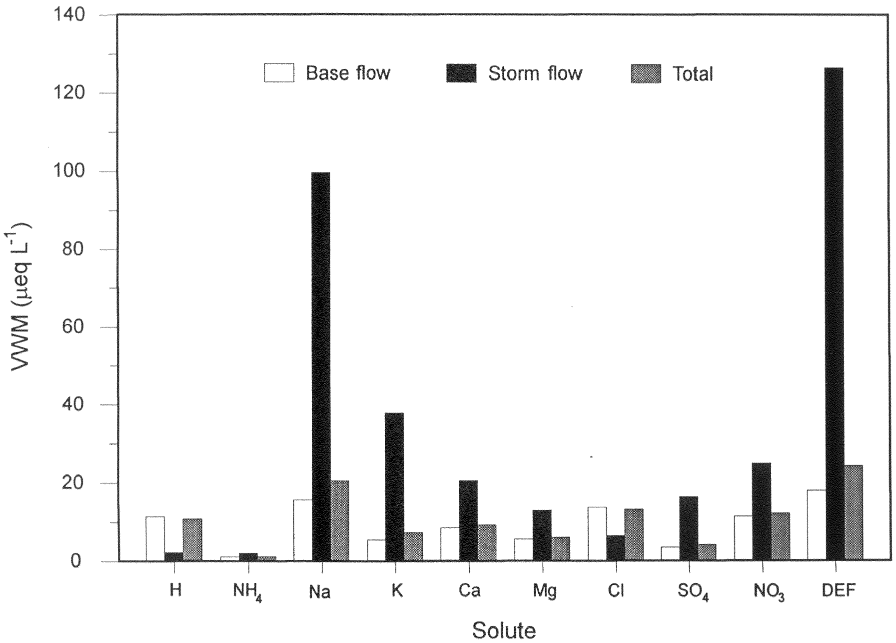

Figure 5. Volume-weighted mean concentrations of selected solutes for base flow, storm flow and total discharge, July 1989 to July 1990. DEF is calculated as cations minus anions.

Particulate export. Particulate export varied considerably, with the largest rate occurring at the LB site (151 kg ha^{-1} y^{-1}). Export during baseflow conditions was responsible for ca. 59% of the total at the LB site. Particulate export from the UB and MB sites was measured during baseflow conditions only, and showed lower export compared to the Lower Braço site (25 and 15% of 89 kg ha^{-1} y^{-1}, respectively; Table 2). Particulate export during stormflow conditions was substantially lower in forested locations (pers. obs.). Discharge-concentration relationships for particulate export had strong counterclockwise hysteresis even during smaller storms, while under baseflow conditions, the export of particulates was independent of discharge (Williams 1993). Although storm flow was responsible for only 6% of the total discharge annually, it was responsible for 41% of the total particulate export in the partially deforested catchment. Rainfall less than 3 mm in size did not have an affect on either stormflow runoff or particulate export.

Discussion

Representativeness of the forested catchment. The streamwater composition data of the adjacent forested catchment (MB) are presented here as a spatial control for the partially deforested catchment (LB). However, the paired-catchment comparison appears pseudoreplicated without pre-disturbance data for MB. Significant pre-disturbance differences in streamwater chemistry of the two catchments are not likely for several reasons. First, a Landsat Thematic-Mapper image from 1984 shows that both catchments above the highwater mark of Lake Calado were fully forested, except for 2 ha of young secondary forest at the lower reach of the forested catchment. Mature forest in the area was qualitatively similar in age and species composition before catchment disturbances occurred, which was confirmed by botanical surveys in both 1984 (Lesack 1996) and 1990 (Williams 1993). Second, the catchments are nearby and have similar elevation, hillslope gradients, perennial stream flow, and Oxisols. These characteristics suggest weathering products and erosion would influence the streamwater chemistry of both catchments similarly because of the similar annual rainfall totals measured in 1984 and 1990. Thus, although the lack of continuity in the data collection is a valid concern, the forested catchment is probably a suitable spatial control.

The most important comparison in this study is that of the partially deforested catchment when it was completely forested (1984) to when it was partially deforested (1990). Possible interannual variability could be falsely interpreted as an effect of disturbance in this comparison. However, given that annual rainfall totals in both studies were similar (2870 and 2754 mm, respectively), potential differences in the chemical composition of streamwater due to either concentration (dry year) or dilution (wet year) effects of variable rainfall could not have had a large influence. Hence, the comparison of the two studies suggests that the differences observed in streamwater solute concentrations in 1984 and 1990 were primarily a consequence of disturbances.

Streamwater solute composition in the partially deforested catchment. Solute mobilization from the upper soil horizons after deforestation is a typical response to disturbance in both temperate and tropical catchments (Uhl & Jordan 1984; Likens & Bormann 1995). The combustion of forest organic matter generally causes an increase in soil pH and alkalinity (Tiedemann et al. 1978). Higher soil alkalinity can reduce Al and Fe solubility (Hecht 1982), and temporary increases in P availability result from soil heating and ash conditions. Thus, increased nutrient availability after slash and burning creates an environment conducive to plant growth (Jordan 1987).

Eutrophication is one possible consequence of increased nutrient fluxes into an aquatic environment and is generally associated with N and P removal from populated areas or areas under intensive land use. Nutrient inputs to adjacent receiving waters can be utilized by phytoplankton, and P is generally limiting in freshwater environments (Schindler 1978). However, Setaro & Melack (1984) showed that Amazon floodplain lakes can be both N and P limited, and this suggests that increases in N and P yields following deforestation could affect phytoplankton biomass.

The streamwater draining undisturbed areas in the Lake Calado basin typically is transparent, acidic (pH <5), and low in dissolved solutes (ionic conductivity $<10\ \mu S\ cm^{-1}$). A comparison of solute yields for the partially deforested catchment under disturbed and undisturbed conditions shows large differences for most solutes (Table 3). By contrast, if the 1990 NO_3^- yield for the partially deforested catchment is adjusted to the 1984 discharge value, NO_3^- export is 3.0 versus 2.7 kg ha^{-1} yr^{-1}, respectively, due to streamwater [NO_3^-] in the partially deforested catchment that was similar to that measured in 1984 for most of the year (Figure 6). Considering the large fluxes of NO_3^- from the rooted zone observed in a cut plot in the partially deforested catchment, high NO_3^- bank seepage associated with areas lacking riparian buffer strips (Williams et al. 1997a), and large percentage of the catchment that was deforested at the time of this study (80%), streamwater [NO_3^-] is lower than would be expected. We suspect that NO_3^- immobilization in the riparian zone by aggrading vegetation and denitrification may reduce N inputs to the stream (McClain et al. 1994; Williams et al. 1997a). Thus, although NO_3^- yields increased over pre-cut conditions in the partially deforested catchment, this was primarily a result of larger discharge as opposed to an increase in streamwater [NO_3^-]. A concomitant increase in NH_4^+ yields was not as apparent as for NO_3^-, and may be a result of sorption and nitrifying activity in the soils and stream that can significantly reduce available NH_4^+ (Jordan 1987).

Phosphate removal from the upper soil horizons was low in comparison to other solutes because PO_4^{3-} can be fixed by Fe and Al in the soil before it is leached. However, TP yields increased since 1984 (Table 3), presumably as a result of deforestation and the increased sediment transport that was occurring due to erosion. The amount of sediment exported from the partially deforested catchment was 151 kg ha^{-1} y^{-1}, with 41% attributed to storm events. This total is higher by a factor of about 5 compared to the mean particulate matter loss for the Hubbard Brook catchment (Likens & Bormann 1995), but is low compared to sediment yields of 100–1,000 kg ha^{-1} y^{-1} observed in disturbed temperate watersheds (Meade 1982). In contrast, our estimate is similar to the median estimate of erosion (181 kg ha^{-1} y^{-1}, n = 16) in the tropics resulting from shifting cultivation (Wiersum 1984).

It was common for the stream in the partially deforested catchment to become turbid during storm events, while the forested catchment stream remained clear. The availability of P in downstream receiving waters

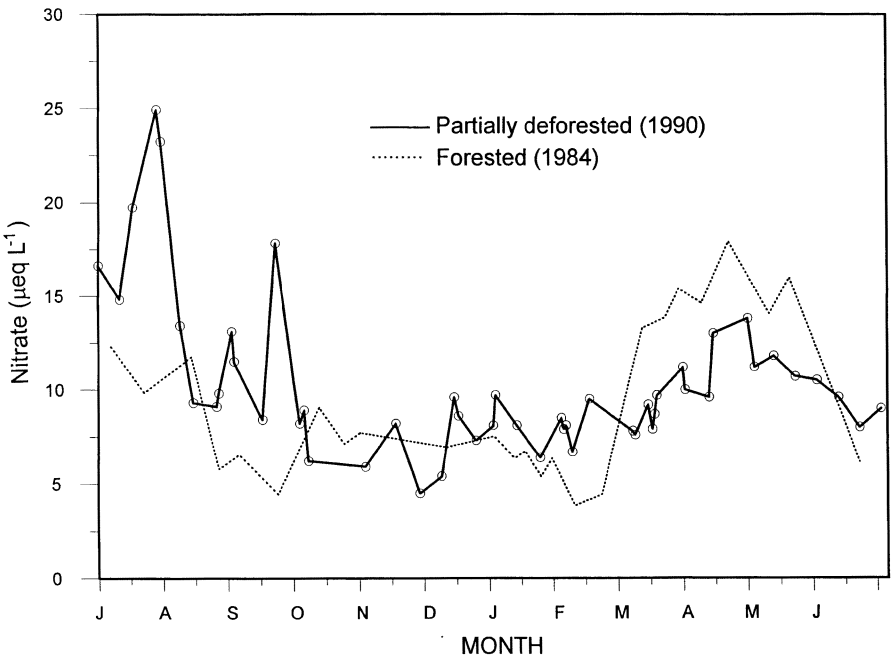

Figure 6. Comparison of streamwater NO_3^- concentrations in the partially deforested catchment stream, July 1989 to July 1990, and February 1984 to February 1985. The 1984–1985 data are presented consecutively from July 1985 to February 1985, and February 1984 to July 1985.

increased due to this added sediment flux. For example, sediments analyzed for PP by combustion (Andersen 1976) had export values that were high compared to those calculated for the forested catchment (0.30 versus 0.01 kg ha^{-1} y^{-1} particulate P, respectively), and storm related transport accounted for 17% of the total (0.05 kg ha^{-1} y^{-1}). The concentration of PP in baseflow sediments was over 3 times higher than that of storm flow (2.17 μmol PP mg^{-1} versus 0.63, respectively), and suggests that particulates from overland flow (dominant in larger storms) are not as high a source of P as the particulates found in baseflow transport, perhaps due to the suspension of coarser materials during larger storm events. Nevertheless, particle movement is the primary mechanism of P removal from the partially deforested catchment. These P inputs to the stream are much higher than previously recorded and indicate that 8 times more P was exported from the study site catchment in 1990 compared to 1984 because of disturbances in the intervening years.

Malmer (1996) observed similar increases in the dissolved and particulate fractions of P in catchments converted to plantation forestry by clear-felling and burning in Malaysia. About 72% of P export was attributed to the

Table 3. A temporal comparison of the yields of surface runoff for the partially deforested catchment in 1990, and in 1984 before cutting (Lesack 1993b). Total phosphorus was calculated by both Valderrama (1981) and Andersen (1976) techniques, the latter indicated by an asterisk.

	Partially deforested	Forested
	1990	1984
	kg ha^{-1} yr^{-1}	
Annual rainfall (mm)	2754	2870
Annual runoff (mm)	2080	1650
Solute		
H^+	0.22	0.39
NH_4^+	0.31	0.18
Na^+	9.87	2.60
K^+	4.82	0.46
Ca^{2+}	3.65	0.54
Mg^{2+}	1.24	0.33
Cl^-	10.50	7.82
SO_4^{2-}	1.46	1.08
NO_3^-	3.64	2.67
TDN	6.44	3.61
TN	9.14	4.31
PO_4^{3-}	0.08	0.02
TDP	0.33	0.05
TP	0.48	0.08
TP*	0.63	

dissolved fraction, whereas in our study, this was about 52%. Malmer indicates that the largest proportion of particle transport occurred during storm flow (Malmer 1990), which is in contrast to the observations in our study that indicate about 60% of the particle transport occurred during base flow. Based on the high proportion of PO_4^{3-} to TDP in his study (61%), Malmer rejects the possibility that overestimation of dissolved P in streamwater occurred due to the inability of fiber filters to remove the finest particles. In contrast, the possibility that colloids contributed to TDP is somewhat higher in our study since the proportion of PO_4^{3-} to TDP is about 13%. Total P export for our study and that of Malmer are 0.63 and 2.22 kg ha^{-1} yr^{-1}; the latter is expectedly higher due to disturbances caused by the mechanical extraction of felled trees.

Williams et al. (1997a) observed that solute fluxes from the rooted zone in cut and burned plots were higher than in forested plots. Uhl & Jordan (1984) observed increased fluxes of K^+, Mg^{2+} and NO_3^- from the rooted zone of a small plot for up to 5 years after cutting and burning in the Venezuelan Amazon. A reduction in solute export from recently disturbed catchments can be attributed to newly aggrading vegetation (Vitousek & Reiners 1975), and riparian buffer strips reduce sediment and solute losses from disturbed areas (Williams et al. 1997a). Hence, elevated solute inputs to receiving waters resulting from deforestation are likely short-term ($<$3 y) in areas that are converted to agricultural plots and subsequently abandoned after several crop plantings.

By contrast, solute inputs to floodplain lakes resulting from the effects of colonization or conversion to cattle pasture in upland catchments and areas bordering Lake Calado are probably longer-term ($>$10 y). For example, periodic domestic use of the stream by colonists contributed to the higher fluxes of solutes measured from the partially deforested catchment; the catchment was inhabited by at least 10 families who used the partially deforested catchment stream as a source of water in October 1994 (pers. obs.). Stream sampling during periods of the day when there was little activity minimized the inclusion of usage effects by colonists in our estimates of solute flux from the partially deforested catchment. Bathing in the upstream reaches was infrequent due to the lack of larger pools, but washing clothes and cooking implements occurred daily. Pools constructed in the stream bank used for soaking manioc tubers were also common, but differences in solute concentrations between pool and streamwater were not significant. Hence, although stream usage by colonists probably had a larger effect on solute export from the partially deforested catchment than our measurements would indicate, periodic inputs of water to the stream from pools used to soak manioc were not likely a strong source of chemicals to streamwater.

Alves (1993) observed significantly higher pH, ANC, NO_3^- and P fractions in the partially deforested catchment compared to the forested catchment from May 1988 to April 1989. These data are mostly consistent with our observations since the partially deforested catchment was about 40–60% deforested in 1988–1989, and much of the deforested area was on steep hillslopes without riparian buffer strips. Unfortunately, short-term catchment studies do not allow us to determine the duration of increased N and P exports to Lake Calado, or to quantify the individual effects of deforestation and colonization.

Overland flow. The increase in streamwater solute concentrations of the partially deforested catchment was partly attributed to overland flow. The presence of older deforested plots situated on steep hillslopes contributed to stormflow runoff via overland flow even during brief, low-volume storms of high intensity. An earlier estimate of the area contributing to overland flow in the partially deforested catchment before it was cut (1984) was 4%, and did not vary during larger storms of up to 100 mm (Lesack 1993b). This area is similar in size to the channel floodplain and suggests that stormflow runoff in undisturbed catchments is generated only when the stream bank becomes saturated (Nortcliff & Thornes 1981). By contrast, stream water of the partially deforested catchment was influenced more by overland flow generated from beyond the floodplain area on hillslopes without riparian buffer strips. During larger rain events, overland flow on steep embankments of deforested areas was common (pers. obs). Although the inputs from overland flow to streamwater discharge could not be quantified in this study, this additional overland flow would be responsible for a higher ratio of stormflow to baseflow runoff at the Lower Braço sampling station after partial deforestation of the study site. However, the stormflow to baseflow ratios in 1990 (6%) and 1984 (5%) are similar, suggesting that stormflow runoff in the present study may have been underestimated.

Possible errors in water balance. The rainfall estimate for our study was calculated using the data collected from 2 pairs of manual rain gauges (Figure 2), which differed by only 4%. Winter (1981) cites estimates of error for daily mean rainfall of 4% at a density of 2.6 km^2 $gauge^{-1}$ and an error of about 5% for point estimates of annual precipitation. Therefore, the expected error in our measurement of incident rainfall over the study period is probably no more than 5%.

The expected error associated with the measurement of annual discharge using a calibrated weir and water level recorder is typically less than 5% (Winter 1981). However, since we extrapolated the discharge from the 1990 weir site to the location of the downstream weir site (1984) by salt dilution, we increased the error associated with the final discharge estimate. The Lower Braço sampling station was suitable for using salt dilution measurements because: 1) an adequate stream length allowed complete mixing of the salt slug, 2) there was a discrete stream channel, and 3) there were no significant inputs between the point of salt addition and measurement. Moreover, we used NaCl as a conservative tracer to prevent absorption effects (Rantz et al. 1983) and had reasonable recovery of the salt slug (total $\pm$8%) on every occasion. Since the error associated with the salt dilution technique under the right conditions can be similar to that of more conventional methods used to measure discharge (Herschy 1985), we are confident that using a combination of salt-dilution, and cross-sectional area and stream velocity techniques to extrapolate to the downstream reach of the partially deforested catchment stream resulted in no more than ±10% error for base flow, and ±25% for storm flow. The latter is attributed to the two independent sources of error in storm flow (i.e. the standard error of the regression coefficients from the rating curve, and the error of the salt dilution measurements). Hypothetically, if the salt-dilution technique is extrapolated to the entire catchment with an error of 20% for base flow and 50% for storm flow, the sum of these errors equals 454 mm, which represents a maximum uncertainty of 22% in the calculation of our runoff coefficient. Moreover, because storm flow is only 6% of the surface runoff measured from the partially deforested catchment, this has a small influence on the error associated with our discharge estimate. For example, if we assume that the errors associated with base flow and storm flow are 10 and 100%, respectively, the sum of these errors equals only 321 mm (15% uncertainty).

Alternatively, the streamwater discharge estimate was calculated by dividing total discharge at the 1990 weir site by the area upstream (about 15 ha). The product of this number (82 mm) and the area above the 1984 weir site gives a total discharge of 1923 mm, which is 157 mm (8%) lower than our other estimate of total discharge. Considering the presumably larger inputs of streambank seepage and overland flow to the stream below the 1990 weir site, an area that was almost 97% deforested, both estimates of discharge calculated in this study are reconcilable.

Although the uncertainty of our stormflow estimate cannot be dismissed, another possible explanation for our high runoff coefficient is the anomalously wet year preceding our study. Mean rainfall for Reserva Ducke (3°8′ S, 60°2′ W), Novo Airão (2°37′ S, 60°57′ W) and Carão (2°55′ S, 60°40′ W) was 3187 mm for the year preceding our study (Departamento Nacional de Água e Energia Eléctrica – DNAEE), whereas that of the study year at these locations was 2721 mm, which is similar to the annual rainfall measured at Lake Calado (3°15′ S, 60°34′ W) in our study. Although there is considerable variability in rainfall amounts at the spatial scale of small catchments, assuming the rainfall amounts at Lake Calado are proportional to the averages of the above locations, the 1989 and 1990 water years are estimated to be about a 1 in 50 wet year (DNAEE) followed by a 1 in 10 wet year (Lesack & Melack 1991). High rainfall during the wet season of 1989 caused record flooding of the Solimões (not shown) and Negro rivers (Figure 7). Record stage of the Negro River in 1989 is not necessarily indicative of high rainfall at Lake Calado, but does suggest that rainfall amounts were high regionally in the wet season of 1989. These data suggest that the 8 month period of heavy rain from November 1988 to June 1989 increased groundwater storage in the partially deforested catchment (Figure 8a). Higher groundwater storage in 1990 than

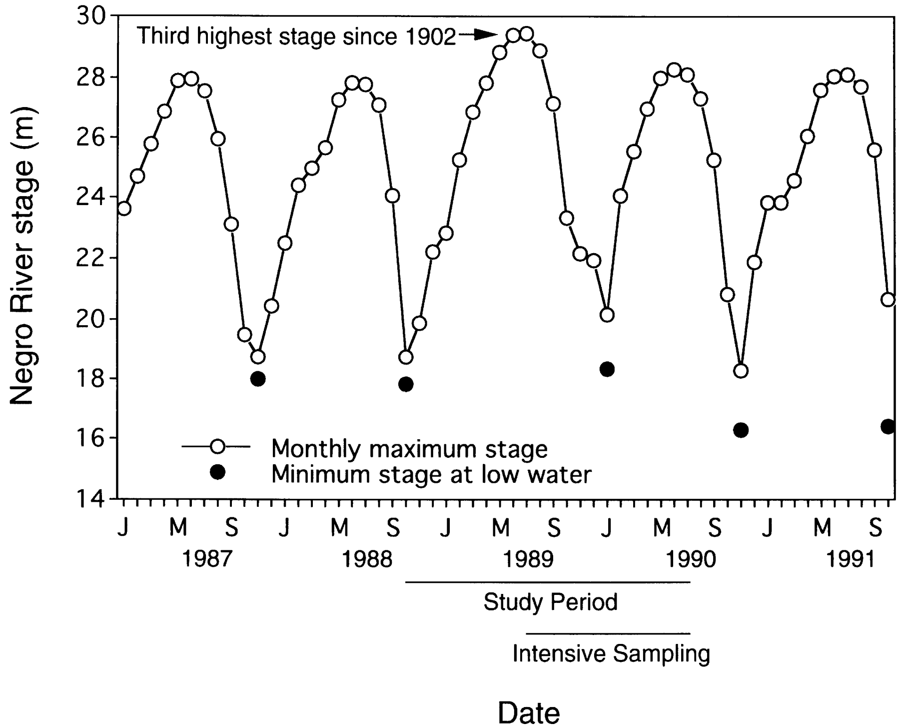

Figure 7. Stage heights at the port of Manaus, Negro River, 1987 through 1991 (Departamento Nacional de Água e Energia Eléctrica – DNAEE). Data are monthly maxima, and the minimum stage of each low water period is indicated.

in 1984 is indicated also by a comparison of the lowest levels of baseflow observed in each study, which were 3 and 0.4 mm d^{-1}, respectively.

The change in groundwater storage as a result of a 1 in 50 wet year preceding a 1 in 10 wet year was calculated using measurements of groundwater height for the 7-month period prior to the phase of intensive sampling beginning in July 1989. A rough estimate was obtained by projecting the declining limb of the base flow curve from January 1990 of our study backwards until the peak in groundwater height was measured in May 1989. A significant relationship ($r^2 = 0.79$, $p < 0.05$; Figure 8b) was determined using instantaneous measurements of groundwater height and mean baseflow discharge from the lowest to the highest streamwater stage periods (i.e. Jan 21–31, 1990 to July 1–11, 1989). Mean groundwater height from the riparian wells of the partially deforested catchment ($n = 4$ at 2 m upslope from the stream bank, Figure 2) were assumed to be representative of changes in streamwater stage because of their close proximity to the stream. Decreasing

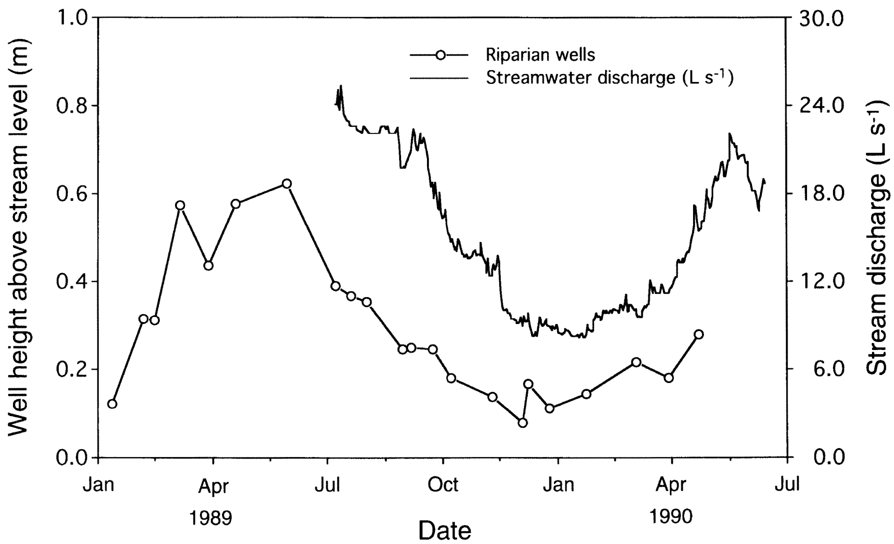

Figure 8a. Well height above stream level and stream discharge for the partially deforested catchment during the study period. Well heights are from 4 riparian wells monitored from January 1989 to May 1990. Streamwater discharge is base flow from the LB site.

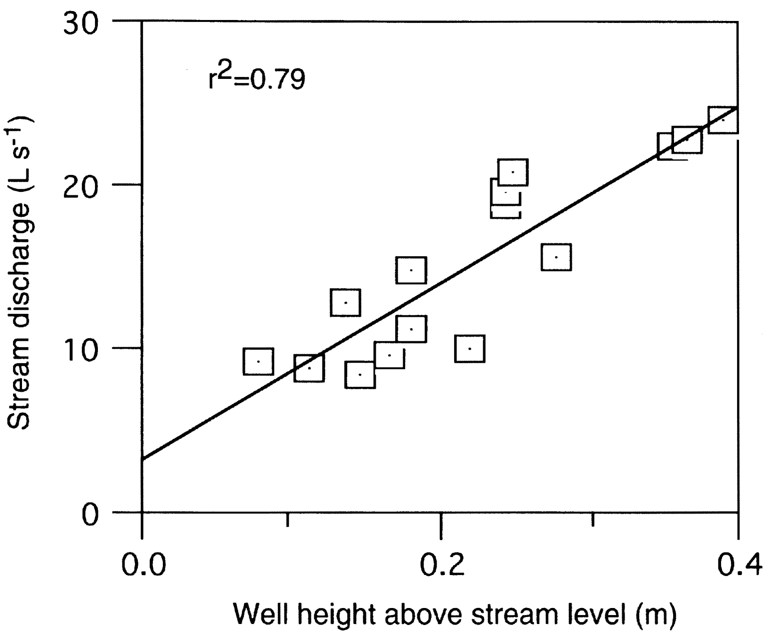

Figure 8b. The relationship of stream discharge versus average well height for the 4 riparian wells in the partially deforested catchment during the intensive sampling period (July 1989 to June 1990).

from the extrapolated measurement of baseflow discharge (i.e. 37 L s^{-1}) to the beginning of our study (21 L s^{-1}) represents an average rate of decline of 0.305 L s^{-1} d^{-1} over 42 days. Assuming that the average discharge rate over the 42 day period was 29 L s^{-1}, the amount of water recovered would be equivalent to about 130,291 m^3, or 557 mm averaged over the catchment area.

The response of streamwater discharge to monthly rainfall shows that streamwater discharge lags behind rainfall by about 2 months (Figure 9). For example, the time it takes rainfall infiltration to move along a presumed pathway from the top slope of the catchment to the stream was calculated. Using a distance of about 75 m (sum of 13 m depth from the soil surface to the watertable during high water, and 62 m to the stream, which was the position of our top groundwater well (Williams et al. 1997a)) and the geometric mean of hydraulic conductivity calculated by Lesack (1993b), indicate that the residence time of water moving along this pathway in the partially deforested catchment would be about 58 days. Given hydraulic conductivities for the catchment that range from $4.6(10)^{-6}$ to $3.7(10)^{-5}$, the relatively large amount of groundwater storage resulting from high rainfall during the 1989 wet season could not have fully drained from the partially deforested catchment by the beginning of our study in July 1989. Hence, some of the residual soil-water from the 1 in 50 wet year of 1989 was responsible for our high estimate of surface runoff and for inflating our runoff coefficient.

Comparison of water balance characteristics with previous work. There are three other catchment scale studies that include estimates of water balance conducted in the central Amazon. Two of these studies are summarized in Franken & Leopoldo (1984) and the third is that of Lesack (1993b), which took place in the same catchment as the present study. All the catchments are similar in forest cover, drainage area (0.234 to 23.5 km^2), and stream order (1° to 3°), but estimates of runoff and residual evapotranspiration (ET) varied considerably (400 to 1650 mm and 1120 to 1675 mm, respectively). It is logical to hypothesize that differences in runoff and ET can be attributed to differences in local vegetation cover, soil characteristics and annual climate. Given that mean annual rainfall in the Amazon basin is 2200 mm (Salati et al. 1979), and that rainfall generally ranges from 2000 to 2800 mm over a 10-year period (Lesack & Melack 1991), runoff and ET should be well correlated to annual rainfall in the central Amazon. Dietrich et al. (1982) and Shuttleworth (1988) argue that the predominant regulatory mechanisms are increased rainfall interception in wet years, which increases ET, and soil moisture deficits in dry years, which have the opposite effect.

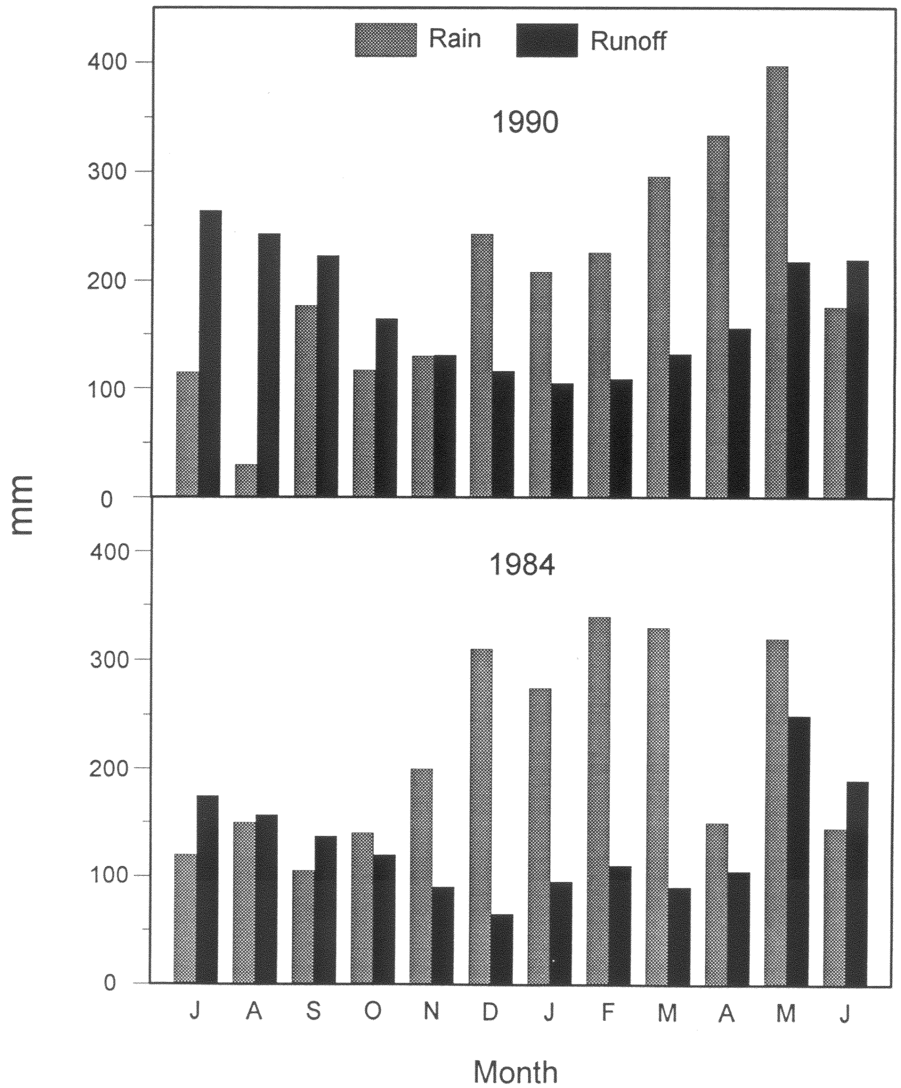

Figure 9. Comparison of monthly rain and runoff for the partially deforested catchment in 1990, and in 1984 when the catchment was completely forested.

However, Lesack (1993b) observed that reconciling the differences in runoff and ET for the above catchments was difficult. In a review of tropical catchments, Bruijnzeel (1990) concluded that ET in tropical lowland forests is similar even among water years with variable rainfall, and that probably the best available estimate of ET is 1320 mm for the central Amazon (Shuttleworth 1988). Because the water budget of Lesack (1993b) was the only study in the central Amazon where residual ET was reconcilable by measurement error with Shuttleworth's estimate, and because the catchment is the same as in the present study, it is the best comparative data set to evaluate potential changes in residual ET after partial deforestation.

Although rainfall in the year preceding our study was a 1 in 50 wet year, groundwater height in the riparian wells during the low-water period was similar in 1989 and 1990 (Figure 8a). Assuming that similar heights in the riparian wells indicate that streamwater discharge was similar during the low-water periods of 1989 and 1990, using the same rational as Lesack (1993b) to calculate soilwater storage for the partially deforested catchment suggests that there was no change in soilwater storage from the low-water period of a 1 in 50 wet year to that of a 1 in 10 wet year. However, this technique does not account for the upslope storage reservoir indicated by measurements of groundwater height in 16 wells distributed in 4 transects within the partially deforested catchment (Williams et al. 1997a). If the catchment is divided into 5 zones representing different groundwater wells upslope from the stream, and we estimate the interstitial water in saturated Oxisols, the difference in groundwater storage from low water 1989 to low water 1990 for each zone of the catchment can be calculated.

We estimated interstitial water volume as the product of the matric potential and the porosity of Oxisols from Fazenda Dimona, about 100 km north of Manaus. Pore space ranges from 50 to 80% in uncompacted Oxisols of the area (Chauvel et al. 1991). Porosity is high, but Chauvel et al. have shown that the pore size distribution is dominated by very fine pores which begin to release water only below a matric potental of -3 m Pa. Considering that there is little influence by vegetation on soil water in deforested areas, the volume of water available for percolation from saturated Oxisols below the water table, or plant available water capacity, is 7% (Correa 1985). Accordingly, the amount of interstitial water that would drain from saturated Oxisols is anywhere from 4 to 6% of the total volume of soil water represented by the difference between the lowest well measurements in January 1989 and those of January 1990. We used the median of this range, or about 5%

The change in soilwater storage in the partially deforested catchment was calculated using the product of the estimate of interstitial water derived above and the mean change in groundwater height in the catchment from low water 1989 to that of 1990. Mean groundwater height was weighted by the area representing different zones of the catchment. We estimated that the top-slope wells represented the largest area of the catchment (about 61%), whereas the mid-slope wells were 33%, and the riparian zone the remainder. For example, the difference from the maximum groundwater height in January 1989 to that of January 1990 in the top-slope wells was about 0.4 m. Since the top-slope wells represent an area of about 14 ha, the total volume of soil and water for this zone is approximately 56,000 m^3. The product of the total volume and our estimate of mobile interstitial water (5%) indicates that 2,800 m^3 more transient soil water was present in the upslope zone of the catchment during low water 1989 compared to that of 1990, or 20 mm distributed over the upslope-well zone using the estimate of interstitial water calculated above. Summing the estimates of soilwater storage from every well zone indicates that there was a total net decrease of about 45 mm in soilwater storage averaged over the catchment from 1989 to 1990.

Our estimate does not account for the change in soilwater storage that resulted from the large rainfall of the latter part of the wet season in 1990. We could not make this calculation since we do not have well or stream discharge data for the low-water period following our study, which is an important distinction in our calculation of soilwater storage compared to that of Lesack (1993b). Generally, a water year extends from one low-water period to another, which constitutes an annual hydrograph; our study includes part of two separate hydrographs. Although soilwater storage decreased from low water 1989 to that of 1990, we suspect that the change in soilwater storage from low water 1990 to that of 1991 would be somewhat larger due to the higher well heights measured in the wet season of 1989, just prior to the beginning of our intensive sampling period.

To complete our water balance, we assumed that Lesack's calculation of subsurface losses in 1984 (about 2.5% of non-evaporative outflow) from the partially deforested catchment is the same in the present study, or 52 mm averaged over the catchment area using our estimate of streamwater runoff for the 1990 water year. The comparison in Table 4 indicates that with similar rainfall occurring in both the 1984 and 1990 studies, streamwater discharge was significantly higher in 1990, whereas subsurface losses were slightly higher, and soilwater storage was lower. Consequently, the estimates of ET calculated by Shuttleworth (1988) and Lesack (1993b) were factors of 2.0 and 1.7 times higher than the residual ET estimate calculated for our study, respectively.

Our results indicate that ET in the partially deforested catchment was significantly lower after it was 80% deforested (1990) compared to when the catchment was completely forested (1984). This finding is consistent with the observation that various types of tropical forest conversions increase water yields (Bosch & Hewlett 1982), although Bruijnzeel (1990) argues that using runoff yield to calculate residual ET is unreliable without accurate measurements of subsurface water loss. Unfortunately, because most studies reviewed in Bosch & Hewlett do not have estimates of subsurface water loss and antecedent rainfall, they cannot be used to accurately infer residual ET

Table 4. A comparison of the water balances for the partially deforested catchment in 1990, and in 1984 before cutting (Lesack 1993b). All estimates are in mm.

	Partially deforested 1990	Forested 1984
Annual rainfall	2754	2870
Annual runoff	2080	1650
Subsurface outflow	52	42
Change in storage	−45	57
Evapotranspiration	667	1121

associated with forest conversion. Moreover, whereas most water balances do not include estimates of groundwater storage, our conservative estimate indicates that variability of groundwater storage can be up to 4% of annual precipitation.

The observations that discharge was higher and ET was lower after partial deforestation of the study-site catchment should be qualified with a proper accounting of the uncertainty involved in our water budget. Assuming equivalent rainfall for the 1984 and 1990 study periods, baseflow discharge was higher by about 425 mm in 1990 compared to 1984. Although higher base flow was presumably a result of the extent of deforestation in the partially deforested catchment, high antecedent precipitation probably inflated our base flow estimate. Additionally, stormflow discharge for the study site was higher only by about 25 mm in 1990 compared to 1984. Because overland flow was observed in cut and not in forested areas, our stormflow estimate may be conservative due to the possible error ($\pm$10 to 50%) involved in the extrapolation from the upstream to the downstream weir site. Nevertheless, because storm flow is only 6% of the total runoff estimated for the partially deforested catchment, large stormflow uncertainty has little effect on our estimate of total runoff. Lastly, residual ET in 1990 was about 450 mm lower than in 1984. Because our estimate of base flow was elevated from high antecedent rainfall, we postulate that residual ET would be closer to 850 mm without a contribution of groundwater from the 1989 wet season. Cumulative measurement error for the 1990 water budget may have contributed also to the low residual ET in our study.

Despite the inherent uncertainty associated with our water budget, results indicate that the higher ratio of streamwater runoff to annual rainfall measured in the partially deforested catchment in 1990 was due to a combination of reduced ET resulting from deforestation and the influence of stored groundwater associated with high precipitation in the wet season preceding this study. Long-term catchment studies incorporating a range of catchment disturbances would assist in quantifying the relationship of runoff and ET to variable rates of rainfall in the central Amazon.

Uncertainty of solute flux estimates. Although baseflow runoff and chemistry have low uncertainties in this study (<10%), those of storm flow and subsurface sources are likely higher. Only two storms were extrapolated to the complete set of storms that occurred during the study period to calculate stormflow solute fluxes. Moreover, subsurface outflow estimates have high levels of uncertainty because of the potential variability of hydraulic conductivity estimates. In contrast, subsurface water chemistry measured in groundwater wells (Williams et al. 1997a) has low uncertainty (Table 5).

The individual errors in solute flux from the partially deforested catchment were propagated to give an estimate of overall error. Propagation of the component errors to obtain an approximate estimate of the overall error associated with the total flux of exported solutes was performed as in Lesack (1993a), and is briefly outlined here. The reduced term of the propagation equation (Reckhow & Chapra 1979; Taylor 1982) is:

$$S_T = (V^2 S_c^2 + C^2 S_v^2)^{1/2} \tag{1}$$

where V is the volume of water, C is the concentration of water, and S is the standard error with the subscripts of volume (*v*) concentration (*c*) and total flux (T). After calculating the standard error of the total flux of base flow (BF), storm flow (SF) and subsurface flow (SUB), the error of the sum of these three components of water flux can be estimated as:

$$S_{total} = (S_{TBA}^2 + S_{TSF}^2 + S_{TSUB}^2)^{1/2} \tag{2}$$

For example, the mean annual concentration of Na^+ was 15.6 μeq L^{-1} in baseflow water (C_{BF}), 103.3 μeq L^{-1} in stormflow water (C_{SF}), and 26.5 μeq L^{-1} in subsurface water (C_{SUB}). The annual volumes of water estimated from the water balance above are 1959, 121, and 52 mm for base flow, storm flow, and subsurface flow, respectively. Sodium has an uncertainty of about 4% (Table 5), which is the standard error computed from Tukey's Jackknife procedure (Sokal & Rohlf 1981), and the error of calculating base flow using our extrapolation method is about 10%. Using Equation (1) to propagate these uncertainties gives an error of about 11% for the flux of Na^+ by baseflow water (Table 6). If it is assumed that stormflow concentration and discharge have uncertainties of 100 and 50%, respectively, Equation (1) gives an error of 112% in the flux of Na^+ in storm flow. Moreover, if it is assumed that subsurface water has an uncertainty of 75%, and the standard error (7%)

Table 5. Comparison of the solute compositions of base flow (BF), storm flow (SF) and subsurface water (SUB). All concentrations are in μM. The standard errors of mean concentrations are derived from Tukey's Jackknife procedure for base flow and subsurface water. Subsurface water was sampled from 16 groundwater wells distributed in 4 transcets in the control and experimental plots (Williams et al. 1997a). Storm flow is assumed to have an error of 100%.

Solute	BF ± SE (1990)	BF ± SE (1984)	SF ± SE (1990)	SF ± SE (1984)	SUB ± SE (1990)	SUB ± SE (1984)
H^+	11.1 ± 0.82	24.5 ± 1.14	2.1 ± 2.1	12.2 ± 6.1	18.8 ± 0.87	3.3 ± 1.7
NH_4^+	1.0 ± 0.12	0.7 ± 0.10	2.0 ± 2.0	1.0 ± 0.5	6.9 ± 1.0	1.4 ± 0.7
NO_3^-	11.7 ± 1.04	11.5 ± 0.97	26.0 ± 26.0	7.6 ± 3.8	24.5 ± 2.5	10.7 ± 5.4
TDN	19.7 ± 1.14	15.3 ± 0.93	62.1 ± 62.1	15.7 ± 7.9	43.5 ± 2.6	12.1 ± 6.1
TN	24.3 ± 0.75	17.2 ± 0.53	147.6 ± 147.6	38.6 ± 19.3	43.5 ± 2.6	12.1 ± 6.1
PO_4^{3-}	0.06 ± 0.003	0.03 ± 0.003	0.1 ± 0.1	0.04 ± 0.02	0.09 ± 0.009	0.05 ± 0.03
TDP	0.48 ± 0.005	0.08 ± 0.007	0.9 ± 0.9	0.22 ± 0.11	0.32 ± 0.028	0.13 ± 0.07
TP	0.61 ± 0.015	0.13 ± 0.013	2.8 ± 2.8	0.64 ± 0.32	0.32 ± 0.028	0.13 ± 0.07
Na^+	15.6 ± 0.68	6.3 ± 0.35	103.3 ± 103.3	13.8 ± 6.9	26.5 ± 1.8	6.8 ± 3.4
K^+	3.9 ± 0.23	0.6 ± 0.02	39.2 ± 39.2	3.3 ± 1.7	4.4 ± 0.07	0.3 ± 0.2
Ca^{2+}	4.0 ± 0.39	0.8 ± 0.14	21.4 ± 21.4	1.0 ± 0.5	4.5 ± 0.92	3.2 ± 1.6
Mg^{2+}	2.2 ± 0.05	0.7 ± 0.05	13.5 ± 13.5	2.7 ± 1.4	0.5 ± 0.03	1.5 ± 0.8
Cl^-	14.7 ± 0.67	13.0 ± 0.79	6.8 ± 6.8	8.3 ± 4.2	23.2 ± 1.4	23.3 ± 11.7
SO_4^{2-}	1.8 ± 0.17	2.0 ± 0.11	17.0 ± 17.0	2.4 ± 1.2	0.7 ± 0.5	1.9 ± 1.0

computed from Tukey's Jackknife procedure for mean [Na^+] in 8 ground-water wells located in a forested plot of the partially deforested catchment (Williams et al. 1997a) is used, Equation (1) gives an error of about 118%. Finally, propagating the uncertainties for the fluxes of Na^+ from discharge and subsurface flows from the partially deforested catchment using Equation (2) gives an error of 32%. Although the total errors attributed to the flux of solutes from all sources range from 12 to 42%, they are much lower than many of the values of uncertainty used in Equation (1) due to the volume-weighted effect of base flow and its relatively low associated error. The high levels of uncertainty used in our estimates of solute flux error are unlikely, albeit possible given the extensive area of the partially deforested catchment that was deforested (80%), the large solute fluxes observed in recently cut and burned plots (Williams et al. 1997a), and the uncertainties associated with our extrapolation of discharge to the downstream sampling station (LB site). Lower uncertainties were calculated by Lesack (1993a) in the partially deforested catchment while completely forested and ranged from about 8 to 19%.

The importance of nutrient export from small catchments to floodplain lake ecosystems. In 1984, much of the Calado watershed was forested, whereas in 1987, about 36% of the lake basin was in agricultural plots or secondary forest (Melack et al. 1992). As of 1990, Landsat Thematic-Mapper images indicate that about 60% of the Lake Calado basin had been cut and converted to agricultural plots or pasture, and the conversion of forested areas in the Lake Calado basin has increased since then (pers. obs.). Because a lake's trophic status and N:P nutrient loading ratios are well correlated with N fixation (Howarth et al. 1988a; b), higher N fixation rates in 1989 than 1980 (Doyle & Fisher 1994) suggest that nutrient inputs from catchment disturbances between 1984 and 1990 may have altered the aquatic ecology of Lake Calado.

A comparison of data from Lesack (1993a; b) and our study provides a plausible scenario concerning the effects of nutrient inputs from disturbed upland catchments to Lake Calado. In 1984 the atomic N:P ratio of stream-water inputs from the forested catchment to Lake Calado was 119 (Table 3), more than 7 times the generally accepted composition ratio of microalgae (Redfield et al. 1963). By contrast, when about 80% of the catchment had been deforested (1990), the atomic ratio of N:P was 32, or twice the Redfield ratio. Since local watershed inputs are an important source of N and P to the lake, increased nutrient export from disturbed catchments may result in altering the trophic status of Lake Calado.

It can be argued that the possibility of eutrophication in floodplain lakes of the central Amazon is unlikely due to the short-term nature of nutrient

Table 6. Comparison of the flux of solutes exported by base flow (BF), storm flow (SF) and subsurface water (SUB) against the total flux exported from the catchment. Export is in moles. Uncertainties used in the calculation of the percent error for each source of water runoff were the following: The errors associated with the concentrations of BF and SUB were determined using Tukey's Jackknife procedure. The error of storm flow concentration was assumed to be 100%. Uncertainties used for BF, SF, and SUB water fluxes were 10, 50, and 75%, respectively.

	BF flux ± SE (%)	SF flux ± SE (%)	SUB flux ± SE (%)	$\sum$ all sources ± SE (%)
Volume of water (m^3)	458400	28300	12200	498900
Solute				
H^+	5084 ± 12	60 ± 112	226 ± 50	5374 ± 12
NH_4^+	458 ± 16	56 ± 112	82.3 ± 121	599 ± 23
NO_3^-	5359 ± 13	728 ± 112	294 ± 119	6387 ± 18
TDN	9023 ± 12	1739 ± 112	522 ± 117	11295 ± 20
TN	11129 ± 10	4132 ± 112	522 ± 117	15794 ± 30
PO_4^{3-}	27.5 ± 11	3.79 ± 112	1.08 ± 119	32.4 ± 17
TDP	220 ± 10	26.2 ± 112	3.84 ± 118	250 ± 15
TP	279 ± 10	79.5 ± 112	3.84 ± 118	363 ± 26
Na^+	7145 ± 11	2892 ± 112	318 ± 118	10361 ± 32
K^+	1786 ± 12	1098 ± 112	52.8 ± 117	2938 ± 42
Ca^{2+}	1832 ± 14	598 ± 112	54.0 ± 126	2485 ± 29
Mg^{2+}	1008 ± 10	379 ± 112	5.98 ± 118	1393 ± 31
Cl^-	6733 ± 11	190 ± 112	278 ± 118	7207 ± 12
SO_4^{2-}	824 ± 14	481 ± 112	8.39 ± 203	1310 ± 42

export from cut areas (Vitousek & Reiners 1975), and the annual flushing that occurs over the rising and falling limbs of the Solimões River hydrograph. However, nutrient export from disturbed catchments can be protracted over a number of years as long as the streams and hillslopes are being utilized by the local inhabitants, especially since our data indicate that P export associated with erosion is the key factor related to the altered nutrient status of Lake Calado. Moreover, during the rising limb of the hydrograph, the Solimões River impounds inputs of water (Lesack & Melack 1995) and nutrients from upland catchments in the floodplain lakes. Doyle & Fisher (1994) observed the highest levels of N fixation at high water. Repeated exposure to higher P inputs over a series of years due to anthropogenic disturbances is one form of cultural eutrophication, and extensive colonization of the Lake Calado basin has exceeded one decade. What impact such nutrient changes may have on the aquatic ecology of floodplain lakes in the central Amazon could not be addressed in our study, and warrants investigation due to the reliance of local and commercial fisheries on floodplain lake environments in central Amazonas.

Conclusions

Results of this study indicate that deforestation has an effect on the hydrochemistry of streams in small catchments of the central Amazon. Streamwater runoff increased due to decreased evapotranspiration in a partially deforested catchment and to regionally high antecedent precipitation. Quantifying the contributions of antecedent precipitation and deforestation to streamwater runoff cannot be achieved without multi-year water budgets, and we stress the need for long-term research on the hydrochemical response of forested catchments to disturbances in the central Amazon and other tropical catchments. Streamwater solute concentrations also increased, but delineating what proportion of the increase was due to either stream usage by colonists or slash-and-burn agricultural practices was difficult. Nevertheless, nutrient ratios in the stream of the partially deforested catchment were significantly altered. Slash-and-burn agriculture on steep grades in close proximity to streams and the continuing colonization of the central Amazon suggest that enhanced soil and solute losses are common. These losses act as nutrient inputs to receiving waters and have potentially important implications to the ecology of Amazon floodplain lakes.

Acknowledgments

The authors wish to thank the Instituto Nacional de Pesquisas da Amazônia (INPA) for its support in Brazil. Funds were provided by the National Geographic Society (4130-89) and the US National Science Foundation (BSR 87-06643). Data analysis was supported by NASA grant NAGW-2602. An earlier version of this manuscript was improved by the comments of S. Hamilton and T. Fisher. Editorial contributions were made by K. Lajtha, H. Elsenbeer, and B. Newton. The manuscript benefited from the valuable suggestions of L. Lesack, and M. Hodnett.

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