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Surface Soil Changes Following Selective Logging in an Eastern Amazon Forest

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ABSTRACT: In the Brazilian Amazon, selective logging is second only to forest conversion in its extent. Conversion to pasture or agriculture tends to reduce soil nutrients and site productivity over time unless fertilizers are added. Logging removes nutrients in bole wood, enough that repeated logging could deplete essential nutrients over time. After a single logging event, nutrient losses are likely to be too small to observe in the large soil nutrient pools, but disturbances associated with logging also alter soil properties. Selective logging, particularly reduced-impact logging, results in consistent patterns of disturbance that may be associated with particular changes in soil properties. Soil bulk density, pH, carbon (C), nitrogen (N), phosphorus (P), calcium (Ca), magnesium (Mg), potassium (K), iron (Fe), aluminum (Al), $\delta^{13}\text{C}$, $\delta^{15}\text{N}$, and P fractionations were measured on the soils of four different types of logging-related disturbances: roads, decks, skids, and treefall gaps. Litter biomass and percent bare ground were also determined in these areas. To evaluate the importance of fresh foliage inputs from downed tree crowns in treefall gaps, foliar nutrients for mature forest trees were also determined and compared to that of fresh litterfall. The immediate impacts of logging on soil properties and how these might link to the longer-term estimated nutrient losses and the observed changes in soils were studied.

In the most disturbed areas, roads and decks, the authors found litter biomass removed and reduced soil C, N, P, particularly organic P, and $\delta^{13}\text{C}$. Soils were compacted and often experienced reducing conditions in the deck areas, resulting in higher pH, Ca, and Mg. No increases in soil nutrients were observed in the treefall gaps despite the flush of nutrient-rich fresh foliage in the tree crown that is left behind after the bole wood is removed. Observed nutrient losses are most likely caused by displacement of the litter layer. Increases in soil pH, Ca, and Mg occur in areas with reducing conditions (decks and roads) and may result from Fe reduction, freeing exchange sites that can then retain these cations. Calculations suggest that nutrient inputs from crown foliage in treefall gaps are probably too small to detect against the background level of nutrients in the top soils. The logging disturbances with the greatest spatial extent, skids and gaps, have the smallest immediate effect on soil nutrients, while those with the smallest spatial extent, roads and decks, have the largest impact. The changes observed 3–6 months after logging were similar to those measured 16 yr after logging, suggesting some interesting linkages between the mechanisms causing the immediate change and those maintaining these changes over time. The direct impacts on soil properties appear less important than the loss of nutrients in bole wood in determining the sustainability of selective logging. Medium-to-low intensity selective logging with a sufficiently long cutting cycle may be sustainable in these forests.

KEYWORDS: Brazil; Logging; Nutrients

1. Introduction

In the last decade, selective logging has become an important land use in the Brazilian Amazon. Most of the Amazon forest was relatively inaccessible until new infrastructure allowed for rapid development and exploitation of forest resources (Uhl et al. 1997). Now, within the Legal Amazon Basin, approximately 16 000–20 000 km² are deforested for agriculture or pastureland, and another 5000–15 000 km² are selectively logged each year (Nepstad et al. 1999). Previous

research on logging impacts in tropical forests focuses on silvicultural aspects, such as damage to remaining trees, regrowth rates, and changes in species composition. Studies are in progress in the Brazilian Amazon to assess the extent and impacts of logging on a regional scale using remote sensing (Asner et al. 2002; Asner et al. 2004a) and eddy flux towers (Saleska et al. 2003; Rice et al. 2004). However, only a handful of the previous or current studies address the potential for changes in biogeochemical cycling to affect the future productivity of these forests or long-term carbon (C) storage (Laurance et al. 1999).

Highly weathered and nutrient-poor soils, oxisols and ultisols, underlay approximately 75% of forested regions in the humid Tropics (Sanchez 1976). Available rock derived nutrients like phosphorus (P), calcium (Ca), magnesium (Mg), and potassium (K) are often in short supply and tightly cycled by the biota (Vitousek and Sanford 1986). Pasture and agricultural productivity in the Amazon is often limited by P and these base cations (Sanchez et al. 1982; Davidson et al. 2004). Phosphorous has been shown to limit tropical forest productivity in Hawaii (Herbert and Fownes 1995; Vitousek et al. 1997), and both P and Ca can limit root growth in some Amazon forests (Cuevas and Medina 1988).

Nutrient losses from logging are primarily through the removal of wood, but leaching losses might also be important (Hedin et al. 1995; Brouwer 1996; Hughes et al. 2000; Martinelli et al. 2000). Estimates suggest that a 30-yr logging cycle for humid tropical forests may be too short for atmospheric nutrient deposition to restore all of the removed and lost base cations, which could result in nutrient depletion to these forests over time (Brouwer 1996; Keller et al. 2004). In addition to nutrient losses, in situ changes to soil structure and nutrient cycling occur following timber harvest. Soil compaction and increased pH are common in more disturbed logging features, such as roads and logging decks, and can persist for over a decade (Congdon and Herbohn 1993; McNabb et al. 1997). These disturbed areas often have higher soil moisture, increased available nitrogen (N) and cations, but declines in total soil C, N, and P (Congdon and Herbohn 1993; Ivo et al. 1996; McNabb et al. 1997). McNabb et al. (McNabb et al. 1997) found that changes in soil bulk density and soil nutrients were still detectable 16 yr after selective logging. They suggested that residual reductions in C, N, and P in impacted areas may be caused by reduced biological inputs caused by persistent soil compaction, while increases in Ca and Mg result from the replacement of dominant species by light-demanding, fast-growing *Cecropia* species that act as Ca and magnesium “pumps.” There are few data available to test hypothesis.

Timber harvests in the Amazon are predominantly selective logging because only a small number of species are merchantable. In the drier eastern Amazon forest, harvests range from 3–20 trees per hectare or 30–200 m³ volume per hectare (Uhl and Vieira 1989; Martinelli et al. 2000; Keller et al. 2004). Reduced impact logging (RIL) is a planned technique with coordinated infrastructure, vines pre-cut, and trees directionally felled (Uhl et al. 1997). RIL can reduce forest damages significantly, particularly for intermediate-scale harvests (Johns et al. 1996; Pinard and Putz 1996; Sist 2000). Ground damage can be up to 50% less and canopy damage up to 75% lower than in conventional logging (Pereira et al. 2001). While the extent of damage may differ, the volume of wood and nutrients removed in timber is independent of logging technique.

After logging some areas have compacted soils with vegetation and surface soils

removed, while others have an open canopy and accumulated coarse woody debris and litter. Previous surveys of biogeochemical impacts have not been spatially explicit. Our objective was to survey in situ changes in surface soil properties and soil nutrients immediately after reduced impact logging in areas with these different localized impacts on forest and ground conditions that are typical in RIL and that can be explicitly linked to the spatial extent of each type of disturbance. We assessed

- 1) changes in litter and forest floor cover,
- 2) changes in surface soil properties and soil nutrients, and
- 3) patterns in these changes across disturbance types.

As a preliminary assessment of possible mechanisms driving or maintaining biogeochemical changes, we also considered how the size of nutrient flushes from fresh tree debris inputs compared to changes in soil nutrients under this debris, whether changes in soils immediately after logging can help us understand changes that persist years after logging, and whether colonizing pioneer or secondary forest species have elevated foliar nutrient content that could feedback to increase soil nutrients over the long-term.

New remote sensing methods are able to detect selective logging (Asner et al. 2004a) and efforts are under way to use them to quantify the extent of logging and hopefully even the different disturbance types (G. P. Asner et al., unpublished manuscript). With these methods, we will be able to scale-up spatially explicit data on changes in soils and improve extrapolation to other areas.

2. Study site

The selective logging site is located in the northern part of the Tapajós National Forest (TNF) in Pará, Brazil, about 50 km south of Santarém, on a plateau that runs along the east side of the Tapajós River at 02°50'S latitude, 55°00'W longitude. The TNF is underlain by old alternating sand and clay lacustrine sediments with the water table located 60–140 m below the surface (Williams et al. 2002). The logging area varies between 150–200 m ASL and is minimally bisected by drainages. The soils are 70% clay dominated (ultisols and oxisols) and 30% sand dominated (ultisols; Silver et al. 2000). Mean annual temperature is 25°C and rainfall averages 2000 mm yr⁻¹ with most inputs falling during the wet season from January through June (Nepstad et al. 2004).

Selective logging was carried out in blocks of approximately 100 ha, and had four spatially explicit impacts (Figure 1):

- logging roads;
- logging decks, where logs are temporarily stacked for movement off site;
- skids trails along which mechanical skidders pull logs from the forest to the decks; and
- treefall gaps, where the canopy falls and remains after being cut from the tree bole.

Each of these is likely to have a different effect on soils.

3. Methods

In March 2003, we sampled four blocks logged between August and October 2002. These blocks were located on clay-dominated (oxisols) soils. We randomly se-

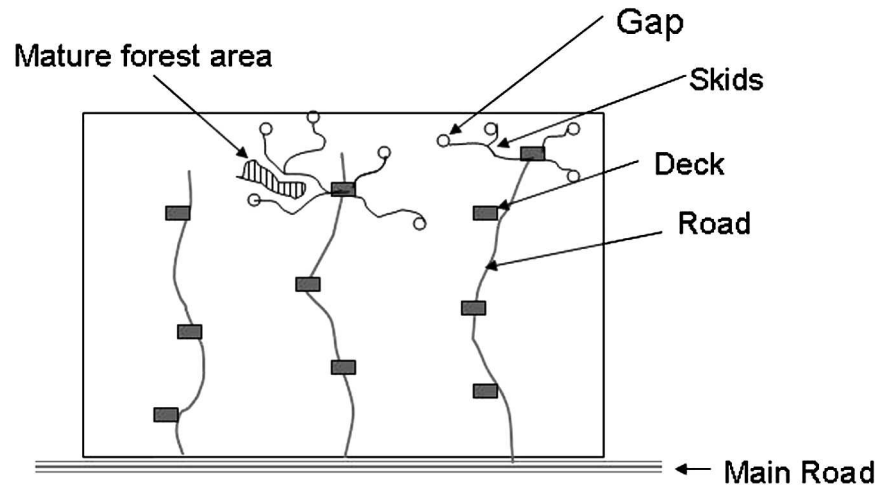


Figure 1. Schematic of a logging block showing different disturbance types.

lected two logging decks for sampling from each block. We sampled the roads leading to the decks every 10 m along a 40-m transect ($n = 5$), and in the decks every 5 m along a central transect (25 m \times 25 m) with three additional samples taken in the center spaced at least 2 m apart ($n = 7$). Out from the decks, one skid was sampled every 10 m along a 40-m transect ($n = 5$). At the first tree crown found, five samples were taken in and around the fallen crown ($n = 5$). In the matrix of unlogged forest at least 10 m from any deck, skid, road, or gap, three to five samples were taken at 10-m intervals along transects ($n = 3$ or 5). Percent cover of the soil surface by litter was measured using point surveys every 10 cm along all 50–100-m transects. Litter in a 15 cm \times 15 cm area was collected down to the soil surface and soils to 10-cm depth were collected at each sampling point. We quantified litter biomass in the different zones. We assumed that changes in soil nutrients would be most detectable in the surface soils where most nutrients essential for biological recycling and plant uptake are held. However, by sampling only the top 10 cm, we could not assess whether movement of C or nutrients to or from lower horizons was important.

Litter samples were dried at 65°C for 5 days and then weighed to determine litter biomass. Soil samples were dried at 70°C for 10 days and subsamples were ground and sieved (2 mm) with charcoal and large roots removed. Total C, N, $\delta^{13}\text{C}$, and $\delta^{15}\text{N}$ were measured on an elemental analyzer mass spectrometer (Delta Plus, Finnigan Mat, San Jose, California) at CENA (University of São Paulo). Exchangeable P, Ca, K, and Mg were measured on 5-g subsamples of soil extracted with Mehlich III (Tran and Simard 1993) and analyzed on the ICP-AES (SpectroFMV-3); pH was measured in water. Two analytical replicates per sample were analyzed using reference material (2710 Montana Soil, NIST) for total soil nutrients.

The P fractionations were conducted on a subset of samples using a modified Hedley fractionation (Tiessen and Moir 1993; Townsend et al. 2002). Three analytical replicates were used for each sample. In short, the soils were subjected to

the following series of extractions: resin strips (in KCl to flocculate soils); bicarbonate (0.5M NaHCO₃); hydroxide (0.1M NaOH); hot concentrated acid (3.6M HCl); and a final peroxide/sulfuric acid digest. This same digest was used on a portion of the bicarbonate, hydroxide, and acid extractions, so that the organic P content could be determined from the total minus the inorganic. We did not include the 1M HCl fraction in our analysis because it mainly contains primary mineral P, which is absent in these highly weathered soils. All samples were analyzed using the Murphy–Riley method (Murphy and Riley 1962) using a spectrophotometer (Shimadzu UV-1203). The resin and bicarbonate extractable pools are the most labile, while the hydroxide, concentrated acid, and residual fractions are likely dominated by mineral (Fe or Al) or organic matter complexed P that varies in recalcitrance (Tiessen and Moir 1993; Cross and Schlessinger 1995). An independent measure of total soil P was determined using the peroxide/sulfuric digest on a replicate soil sample. The sum of all fractions was compared to this independent measure to provide a measure of confidence in the fractionation scheme.

Fresh foliage was collected from 192 upper-canopy species at a site 17 km from the logging area. The species were selected based on relative abundance and comprised all genera listed by Keller et al. (Keller et al. 2001), Nepstad et al. (Nepstad et al. 2004), and Asner et al. (Asner et al. 2004c). Foliage was shot from full sunlight positions using a shotgun, captured and pressed for subsequent drying at 70°C for five days. Samples were transported to the University of Brasília for nutrient analysis using an ICP mass spectrometer (ICP-AES, IRIS/AP, Jarrell-Ash Corp., Franklin, Massachusetts). Nitrogen concentration was measured by Nessler colorimetric method. Two analytical replicates per sample were analyzed using reference material (1547, Peach Leaves, NIST). Data were then partitioned by mature and pioneer habitat, as described by Parrotta et al. (Parrotta et al. 1995).

4. Data analysis

Data for litter biomass and many soil nutrients were not normally distributed, so all data were analyzed nonparametrically, with significant differences among all medians determined using the Kruskal–Wallis rank test ($\alpha = 0.05$; JMP IN 1995 software, SAS Institute Inc.). Where significant differences among groups were found, we used Dunnett's test (JMP 1995 software, SAS Institute) to compare each impact type to the control unlogged forest ($\alpha = 0.05$). Litter biomass data were for two of the four blocks. For P fractions the median and range are shown (JMP IN 1995 software, SAS Institute). The sum of P fractions and the independent measure of total soil P were compared for each impact type using a nonparametric signed rank test ($\alpha = 0.05$).

We estimated foliar nutrient inputs from living tree crowns left in treefall gaps using data on crown structure acquired from the Tapajós Forest and a similar forest at the Fazenda Cauaxi also in Pará (Asner et al. 2004b; Nepstad et al. 2004). An average leaf area index of 5.8 and specific leaf area of 14.3 m² kg⁻¹ are combined with an average canopy diameter of 15.8 m to calculate the foliar biomass in the crown of a typical logged tree. Since merchantable timber are mature forest trees, the canopy diameter used is based on the assumption that half of the logged trees

are dominant and the other half superdominant. The specific leaf area used is for upper-story trees. Foliar nutrient data for mature nonpioneer forest trees collected from undisturbed areas of the Tapajós forest and presented in this paper were used to calculate the nutrient content of an average tree crown for a logged tree. We assume the average area of ground disturbed where these crowns fall and reside after logging are approximately the same size as the canopies themselves.

5. Results

5.1. Litter and soils

Litter cover varied substantially between impact zones 4–6 months after logging. In the unlogged forest and gaps, no bare ground was exposed and skids had only ~5% exposed soil, whereas roads and the center of decks had over 70% exposed soil. Litter biomass in gaps and skids was not significantly different from what was observed in the forest, but deck centers and roads were, both with around 90% less litter (Figure 2). In the decks, most of the litter was found in debris piles around the outer edges. We sampled during the wet season and found areas of standing water in deck centers and along the ruts of many roads; the soils often experienced reducing conditions and Keller et al. (Keller et al. 2005) found elevated nitrous oxide and methane soil emissions indicating anaerobic conditions in these areas. In the roads and decks, upper-horizon soils were often mixed, with subsurface soils laying over surface soils. Mixing is likely caused by bulldozers clearing and leveling these areas. In gaps and skids, bulk density was similar to that of the unlogged control, but in the decks, it was significantly higher, increasing from ~1.04 to 1.25 Mg m⁻³ (Figure 3a; R. C. de Oliveira Jr. and M. Keller 2003, personal communication). Bulk density of the roads was very similar to that of decks. Soil pH was also significantly higher in roads and decks than in the unlogged forest, increasing from around 3.6 to over 4.0 (Figure 3b). It is therefore not surprising that decks and roads showed large changes in soil nutrients as well as soil properties relative to unlogged areas. What is surprising is that they often showed different changes.

Despite the compaction, increased pH and wet conditions, no changes in the concentrations of Fe and Al were observed (Figures 3c and 3d). Soil C and N concentrations in decks and roads were less than in the unlogged control areas, but this was only significant for the roads; the decks had greater variability (Figures 3e and 3f). On average, the roads had 8.0 mg C and 0.6 mg N per gram of soil less than the unlogged forest soils. While there were no significant changes in $\delta^{15}\text{N}$ across damage type, soil $\delta^{13}\text{C}$ was significantly greater in the skids, roads, and decks (Figures 3g and 3h). The concentrations of exchangeable K, Mg, and Ca in soil were relatively constant across damage types except for in the decks, which had significantly higher concentrations of all three (Figures 3j–l). Mehlich III exchangeable P showed the same pattern as total soil C and N; the roads had significantly lower Mehlich III P than the unlogged control areas with a drop of 2 μg P per gram of soil (Figure 3i). Assessing changes in nutrient stocks per volume of soil emphasized increases in exchangeable K, Mg, and Ca observed in the compacted deck soils, while eliminating changes in C, N, and exchangeable

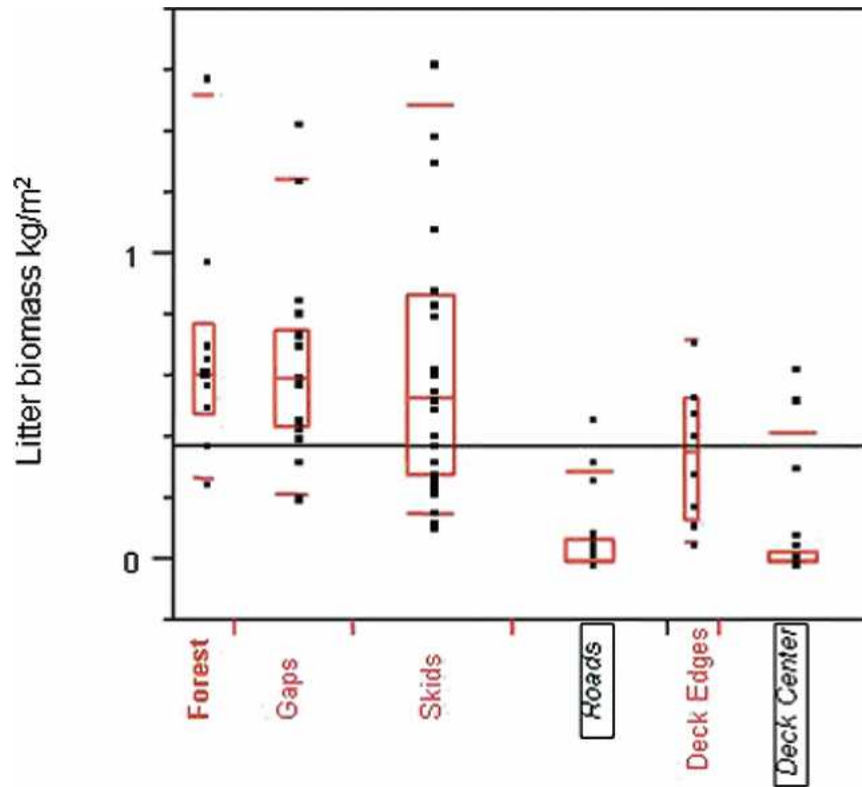
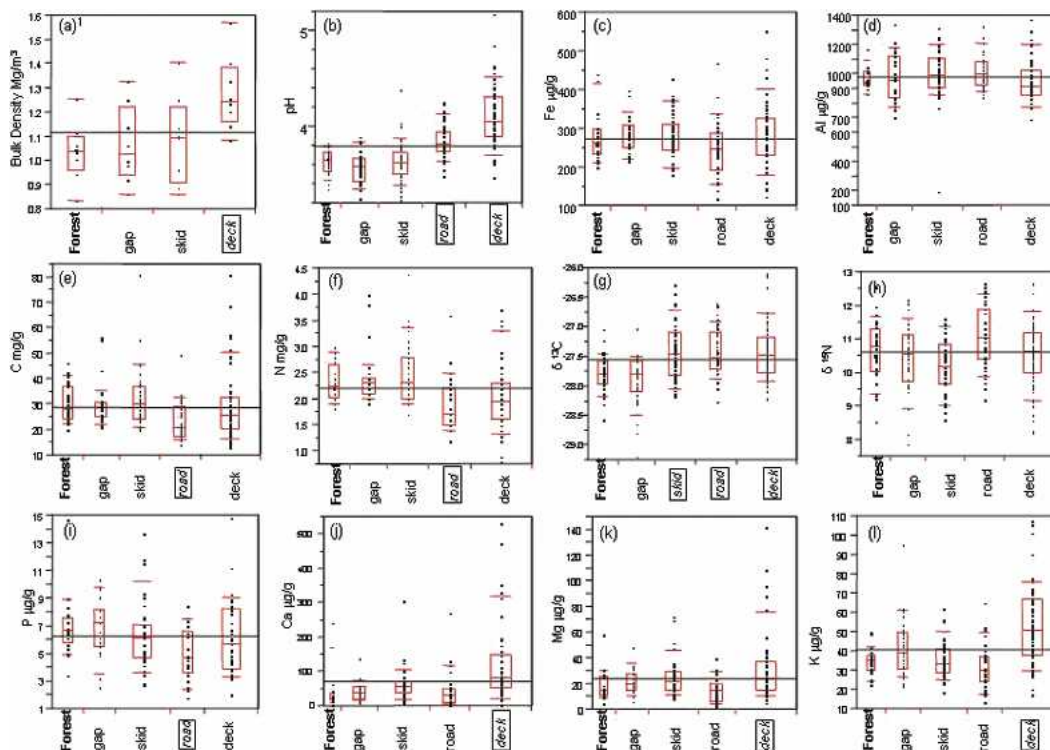


Figure 2. Changes in the distribution and biomass of litter after selective logging in the Tapajós National Forest. The x axis is proportional to sample size; the quantile plots show the data points with the median in the center and 25%–75% of data in the box. The upper and lower 10% of data are designated with dashes, and the feature names with boxes around them have soil properties significantly different than the unlogged control. The horizontal line is the mean of data for all groups combined for comparison across groups.

P concentrations observed in the roads (data not shown). The median K content increased from 35 in the forest to 59 kg m^{-3} in the decks, Mg from 11.2 to 24.5 kg m^{-3} , and Ca from 0 to 69 kg m^{-3} .

Exchangeable P measured in the Mehlich III extraction can provide a rough estimate of changes in labile inorganic (i) P, but P fractionation provides a much more detailed description of soil P including the organic (o) pools, which are considered more biologically available (Table 1). The most mobile P fraction, resin extractable P_1 declined with disturbance intensity, especially in the decks, but these changes were not significant. Similarly in the other inorganic fractions, a decline in the more disturbed road and deck areas was often observed, but not significant. Changes in the organic pools may be more biologically relevant. The roads had about 40% less bicarbonate P_o than the unlogged forest. This is a relatively labile pool that includes microbial P. For the hydroxide P_o fraction, which includes less labile Fe and Al bound P; skids, roads, and decks all had about



Data are from Keller et al. (in press).

Figure 3. Soil characteristics after selective logging in the Tapajós National Forest, Pará, Brazil: (a) bulk density, (b) pH, (c) exchangeable Fe, (d) Al, (e) total C, (f) total N, (g) $\delta^{13}\text{C}$, (h) $\delta^{15}\text{N}$, (i) exchangeable P, (j) Ca, (k) Mg, and (l) K. The x axis is proportional to sample size, the quantile plots show the median with 25%–75% of data in the box and the upper and lower 10% designated with dashes, and the feature names with boxes around them have soil properties significantly different than the unlogged control. The horizontal line is the mean of data for all groups combined.

40% less than the unlogged forest. Changes in the more recalcitrant HCl fractions were not significant. These changes in fractions were not detectable in the sum of all pools, but our independent measure of total P did show the roads had significantly less total P than the unlogged forest. The roads were the only impact zone in which the independent measure of total P was significantly different than that of the sum of fractions ($p < 0.02$; Figure 4).

5.2. Foliar nutrients

On average foliage in the Tapajós forest is high in Ca and low in P (Table 2). The pioneer species have higher concentrations than mature forest species of all nutrients except K. During logging, the crowns of felled trees are left in the gaps to decompose. The live leaves on the crowns constitute a flush of nutrients. The

Table 1. The median and range for phosphorus fractions in $\mu\text{g g}^{-1}$ soil for each logging impact type. Bic is bicarbonate, Hyd is hydroxide, HCl is the concentrated hydrochloric acid, Sum is the sum of all fractions, and Independent is an independent analysis of total P for comparison to the sum. The asterisks (*) show where the P fraction in that impact type was significantly different from the control unlogged forest with an alpha level of 0.10.

Use	N	Resin P _i	Bic P _i	Bic P _o	Hyd P _i	Hyd P _o	HCl P _i	HCl P _o	Residual	Sum	Independent
Forest	6	12.8 (5.2)	6.3 (2.6)	5.7 (3.3)	17.6 (6.8)	25.0 (27.0)	68.2 (19.5)	0 (8.0)	129.5 (108.6)	255.9 (136.8)	296.2 (229.5)
Crown	7	9.7 (10.2)	5.6 (2.7)	3.9 (3.7)	16.8 (4.6)	14.2 (9.1)	60.9 (11.6)	4.4 (13.8)	128.5 (126.6)	246.2 (113.5)	235.5 (114.5)
Skid	6	8.0 (17.9)	6.3 (5.1)	3.8 (6.6)	19.7 (7.1)	13.3* (11.6)	62.8 (5.0)	0 (9.2)	143.1 (81.85)	259.5 (96.1)	254.4 (157.0)
Road	7	9.5 (18.8)	5.4 (7.2)	4.2* (3.8)	17.7 (13.1)	13.2* (32.4)	56.4 (23.3)	1.7 (14.0)	204.4 (123.8)	317.8 (141.2)	216.4* (130.4)
Deck	9	8.7 (9.3)	4.6 (6.8)	4.0 (7.2)	15.9 (16.6)	12.3* (20.7)	63.0 (32.2)	8.0 (32.2)	146.5 (132.0)	265.8 (196.4)	278.4 (95.3)

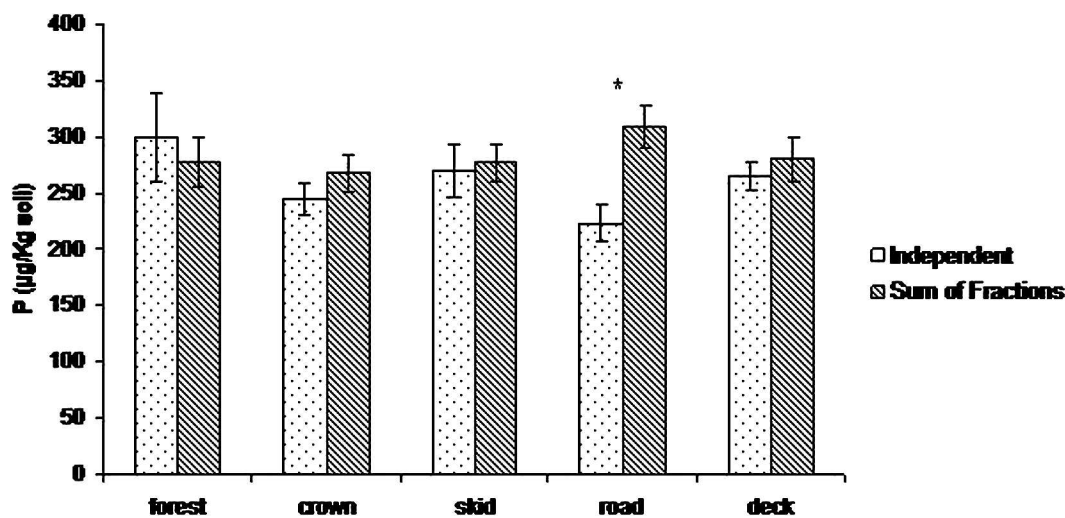


Figure 4. Efficiency of P fractionation determined by comparison of sum of fractions to independent measure of total P. The asterisk (*) indicates a significant difference at the 0.05 level.

average foliar nutrient content of fresh foliage was over 2 times higher than that of typical litter inputs (Table 2). If we use the average crown diameter, leaf area index, and specific leaf area for mature trees, we estimate the foliar biomass transferred to the ground in each gap to be around 6.4 kg of dry leaf biomass per canopy. Based on average foliar nutrient data for mature forest trees, when the nutrient inputs are spread out over the area of the gap, they are 2 to 4 orders of magnitude less than the soil nutrients found in the top 10 cm of soil, suggesting that it would be very difficult to detect their addition (Table 3). For example, a total of 0.15 kg of Mg comes from a single tree crown, but when spread out over the crown area, this is only 0.009 kg m⁻², which is likely undetectable against the background 14.6 kg m⁻² to 10-cm depth of exchangeable Mg in these soils. Comparison of fresh litter nutrients to that of living foliage suggests that trees retranslocate about 70% of foliar P, about half of foliar N and Ca, and less K and Mg.

Table 2. Mean percent of foliar nutrients and fresh litter nutrient content after retranslocation with standard error in parentheses for trees in the Tapajós National Forest, Pará, Brazil.

	Live foliage				Fresh litter
	All	Mature spp.	Pioneers spp.	<i>Cecropia</i>	All
No.	192	171	21	1	32
N	2.03 (0.06)	1.98 (0.06)	2.81 (0.11)	2.89	1.02 (0.09)
P	0.73 (0.03)	0.71 (0.03)	0.88 (0.05)	1.21	0.21 (0.03)
Ca	9.05 (0.55)	8.93 (0.66)	11.69 (0.68)	13.25	4.19 (0.10)
Mg	2.32 (0.16)	2.34 (0.18)	2.88 (0.36)	2.08	1.61 (0.05)
K	5.18 (0.48)	5.27 (0.56)	4.92 (1.15)	9.7	2.08 (0.11)

Table 3. Average soil nutrient content in kg m⁻² to 10-cm depth as compared to estimated foliar inputs from tree crowns left in the gaps.

		C	N	P	Ca	Mg	K
		(kg m ⁻²)	(kg m ⁻²)	(g m ⁻²)	(g m ⁻²)	(g m ⁻²)	(kg m ⁻²)
Soils	Forest	30.92	2.38	4.40	17.28	12.06	30.69
	Gap	28.38	2.25	3.67	28.30	14.59	35.79
Foliar flush	Canopy foliage (kg per canopy)*	3.212	0.130	0.047	0.332	0.149	0.580
	(kg m ⁻²)	0.203	0.008	0.003	0.021	0.009	0.037

* Based on average foliar nutrients for mature forest species and assuming 15.8-m² crown diameter (Asner et al.).

6. Discussion

We observed several changes in soil properties and nutrient availability after logging. Along roads, litter was removed, soils were compacted, pH increased, and total C and N and organic P pools declined. While the logging decks also had litter removed and compacted soils, changes in C and N were more variable and not significant. We can conceive of few mechanisms that would reduce soil concentrations of C and N within 6 months of logging. One possibility is the loss or displacement of litter and surficial organic matter during logging. We suggest that during road construction, litter and surface soils are mostly removed from the road, while in creating decks, soils are reworked in place, leaving more surface soils in the same location but mixed among horizons. Another factor might be erosion during large rain events. Erosion is more likely to occur in roads than decks, but it is probably small from both. Slopes are shallow and the soils are generally high in clay and not very mobile. An alternative explanation for the difference observed between roads and decks is that roads were created before decks and have been leached and eroded over a longer time. Removal of the surface layers would leave behind subsoils that are lower in C, N, and organic P as we have observed. It would also help explain the observed increase in $\delta^{13}\text{C}$. Since $\delta^{13}\text{C}$ increases sharply with depth in the top 50 cm of soil, where topsoil was removed the remaining soils would be enriched in $\delta^{13}\text{C}$ relative to areas where topsoils were not removed (Telles et al. 2003). Leaching alone would probably not create $\delta^{13}\text{C}$ -enriched surface soils. Skids may also have litter and top soils removed, as trees are dragged out with skidders, causing the changes in hydroxide P_o and $\delta^{13}\text{C}$ that are similar to those observed in the roads. A final factor that may affect soil nutrients is a change in biological inputs and uptake. Organic inputs are lowest on the roads and decks, but biological uptake may also be low in these compacted soils. However with rapid regrowth, nutrients must be taken from the soils to replace those lost in the wood biomass removed. If there is a decline in input while uptake is maintained or increased, we would see a decline in C, N, and P as well as other nutrients. This imbalance is unlikely to be measurable immediately after logging.

None of these mechanisms can explain the observed increases in Ca, K, and Mg in the decks and roads. A biological explanation does not seem sufficient. If the compacted soils caused or triggered root decomposition and microbial lysis releasing a flush of nutrients, while biological uptake declined, we would expect C, N, and P to have increased as well as the cations. Instead, we measured declines

in those nutrient pools. A better explanation may be geochemically based. In acidic iron-rich soils, anoxic conditions often result in the reduction of Fe^{3-} to Fe^{2-} and with that comes an increase in pH. The higher mobility of Fe^{2-} causes the dissolution of iron coating around soil particles, freeing exchange sites for other cations. This process may explain the increased retention of exchangeable Ca, K, and Mg (McBride 1994). We saw no significant changes in exchangeable Fe because drying of soil samples before nutrient analyses allows reducing conditions to reverse and Fe to rapidly reoxidize. While increases in exchangeable Fe disappear, changes in pH and Ca, K, and Mg remain. The only explanation we have for why roads do not have elevated cation concentrations like decks is that losses in surface soils are greater than any increases on exchange sites in the 3–6 months after logging. Other logging studies also found elevated soil moisture, Ca, Mg, and K in compacted soils (Ivo et al. 1996; McNabb et al. 1997). These studies hypothesize that biological process such as reduced microbial biomass and root uptake are likely causes for the increases in these cations. The higher concentrations of ammonium and nitrate observed in these studies provide additional support for this explanation. If there is an increase in organic inputs through root or microbial death, the biologically released cations may be bound to the newly available exchange sites.

A reduction of Fe in the compacted soils would also, at least temporarily, increase available P (McBride 1994). Since the soils were dried before analysis, along with the reoxidation of Fe, available P would have rapidly resorbed. The most pronounced changes in soil P measured were the reductions in soil organic P pools in the compacted roads and decks. The surface litter and soils that are removed are a major source and sink for organic P. It is also likely that the unfavorable conditions in compacted soils cause microbes and roots to die and release nutrients. Biologically held organic P is likely released during logging disturbance, with some then sorbed into geochemical soil pools (Garcia-Montiel et al. 2000; Townsend et al. 2002; Asner et al. 2004c). Although we observed no significant increase in the inorganic pools where this sorbed P should be found, there was a trend of increasing P in the residual P pool. The changes in the sorbed pool are likely to be small relative to the inorganic pools and the significance of any change could be lost in the fairly large error prevalent in these measurements.

In previous studies of forest gaps, some have observed increases in nutrients and soil moisture, while others observed little change (Vitousek and Denslow 1986). We found no significant changes in soil carbon or available nutrients in skids and gaps despite a flush of fresh leaf input, likely root mortality, and reduced biological uptake all within the last 3–6 months (Ostertag 1998). Even if it is biologically important, the addition of fresh leaf litter from an entire tree crown is a small addition relative to the background soil pools and thus too small to detect in soils. The branches and remaining trunk of the tree will add even less nutrients, and in fact may result in the immobilization of some nutrients for decomposition.

In the same forest where we studied logging, McNabb et al. (McNabb et al. 1997) assessed changes in soil nutrients 16 yr after selective logging and found that increased bulk density persisted over much of the logged area, with slightly lower N, P, K, and C, and significantly greater Ca and Mg in more disturbed areas. We saw a similar pattern in areas recently logged and hypothesize that the same mechanisms important in short-term changes in nutrients may play a role in main-

taining the long-term changes they observed. For the reduction in C, N, and P after logging, loss of the surface organic matter is the most likely cause of rapid change. To explain how these reductions persist, McNabb et al. (McNabb et al. 1997) suggest lower root biomass in the compacted soils leading to lower organic inputs, or reduced aeration decreasing mineralization of P and K. These reductions in input would have to coincide with maintained or increased uptake of nutrients by trees and microbes. Increased uptake is plausible as trees rapidly regrow and use soil stores to replace nutrients removed in logs. These processes would be set in motion immediately after logging, but may only result in measurable changes years later.

The observed increase in Ca and Mg observed immediately after logging most likely results from reducing conditions in compacted areas freeing exchange sites to retain Ca and Mg. McNabb et al. (McNabb et al. 1997) hypothesize that an increased abundance of *Cecropia* spp. in the disturbed areas creates a long-term enhancement of litter nutrient inputs leading to the persistence of high Ca and Mg. There is evidence of increased pioneer species abundance after logging (Silva et al. 1995), and greater foliar Ca concentrations in pioneer species (Table 2), but *Cecropia*-dominated forests have some of the lowest decomposition rates observed in Amazonia (Rita De Mesquita et al. 1998). It is clear that changes in bulk density and pH are also persistent for years, and we suggest that geochemical mechanisms may remain important in controlling nutrient availability. If compacted soils continue to experience reducing conditions, open exchange sites may continue to capture and hold Ca and Mg inputs. If Ca and Mg inputs from pioneer species do increase, it would accentuate this pattern. Also regional fires may increase local Ca and Mg inputs in precipitation, which might be held and accumulate in the surface soils if exchange sites are available.

Many changes in soils observed after logging are similar to those observed after forests are converted through slash and burn. Although burning is suspected to be the primary cause of elevated pH and increased cation retention (McGrath et al. 2001), it looks as though soil compaction and reduction in logged areas can result in many of the same changes.

Four to six months after logging, and three months into the wet season, our detected changes in surface soils suggest physical, chemical, and biological processes have altered nutrient availability. Many of these changes can persist for years even though some of the mechanisms maintaining these changes may shift over time. One important question for the long-term response of these forests to logging is whether any of these changes in soils could lead to long-term decreases in nutrient availability. Although increases in gaseous losses of nutrients can be substantial in small highly impacted areas, overall fluxes are not large enough to affect soil pools, even though they may be large enough to affect the global warming potential of the forest (Keller et al. 2005). The magnitude of increased leaching losses from logging is uncertain. In a comprehensive study of selective logging in Guyana conducted by Brouwer (Brouwer 1996), with a variety of different measures and estimate parameters, he found that leaching losses were at most 30% of losses due to wood removal. In the Amazon with soils and roots 30+ m deep, we suspect that leaching of nutrients to lower horizons occurs and lateral movement and losses are possible, but losses from the rhizosphere are probably very small (Nepstad et al. 1994, 2001). Although perhaps significant in certain

impact zones, all of these nutrient losses combined integrated over the entire logged area are not large enough to be detected as a decrease in the surface soils of the forest. Nutrient increases, particularly Ca and Mg, appear to be the best indicator of logging in the recent past and for up to 16 yr following harvest (McNabb et al. 1997). Significant changes in soils occurred only in the decks and roads, which when combined, only cover approximately 1.7% of a logged area (Pereira et al. 2001).

By far, the greatest nutrient losses from logging are in the wood that is removed. For the Tapajós National Forest typically three to four trees are removed per hectare (Keller et al. 2004). With this modest harvest rate, and using rainfall nutrient data from a remote site in Amazonas state located 80 km west of Manaus near Lake Calado, it is estimated that within 30 yr, lost nutrients would be more than replaced by rainfall inputs for all except Mg (Keller et al. 2004; Table 4). Other rainfall nutrient data from Markewitz et al. (Markewitz et al. 2004) collected in Paragominas (Pará), the same federal state as this logging study, also show that 30 yr of rainfall will be sufficient to replace nutrients at this harvest intensity, and in this case, even Mg input is sufficient. However, this assertion, that nutrient levels recover within 30 yr, does not hold at higher rates of timber harvest. Martinelli et al. (Martinelli et al. 2000) estimated nutrient losses from a 9-tree-per-hectare harvest near Porto Velho and found much greater nutrient losses. If set to the same 30-yr criteria, rainfall inputs may be insufficient to replace any of these nutrients depending on which rainfall dataset is used (Table 4). The prevalence of fires may in part explain differences in rainfall nutrients. Markewitz et al. (Markewitz et al. 2004) collected bulk precipitation in an area where dry season fires are frequent, while the data in Keller et al. (Keller et al. 2004) are from a more remote area less likely affected by regional land use change. Nutrients in the bole wood of the mature harvest trees vary across the Amazon, nutrients in rainfall may also be affected by large-scale regional forcings, but more localized and seasonal burning may have a large effect. This variability in inputs and outputs makes generalization of sustainable harvest intensity difficult.

Table 4. Nutrient losses in bole wood for a moderate (Mod) and a high-intensity (High) harvest, rainfall nutrient inputs measured in two different locations, and time for rainfall inputs to replace the loss of bole wood nutrients using both rainfall datasets.

Element	Nutrients in moderate harvest ^a (kg) 3–4 trees ha ⁻¹	Nutrients in high intensity (kg) 9 trees ha ⁻¹	Rainfall inputs ^a (kg ha ⁻¹ yr ⁻¹)	Replacement time (yr)		Rainfall inputs ^c (kg ha ⁻¹ yr ⁻¹)	Replacement time (yr)	
				Mod	High		Mod	High
N	56	200	4.16	13.5	48.1	4.0	14	50
P	1	3	0.34	2.9	8.8	0.03	33.4	100
K	14	50	0.9	15.5	55.5	5.0	2.8	10
Mg	9	20	0.3	30	66.7	1.2	7.5	16.7
Ca	18	170	1.32	13.6	128.8	3.2	5.6	53.1

^a Keller et al. (2005); logging in Tapajós National Forest, Pará; rainfall data from Amazonas (Williams et al. 1997).

^b Martinelli et al. (2000); logging in Porto Vehllo, Rondônia.

^c Markewitz et al. (2004); rainfall data from Paragominas, Pará.

Logging is likely to cause subtle changes in soil structure and nutrient dynamics that are detectable both immediately after logging and for years to come. However, these changes do not indicate any substantial loss of nutrients as compared to the soil stores. As suggested by estimates of nutrient losses with log removal, these losses may be replaced by atmospheric deposition (Keller et al. 2004), but could become critical with frequent repeated logging, or high extraction rates (Martinelli et al. 2000). It is possible that nutrient losses could begin affecting forest productivity and still be too subtle a change to be detected in soil nutrients. Measurable increases in Ca and Mg in the surface soils may facilitate regeneration and rapid regrowth after logging.

7. Conclusions

This study is a preliminary survey of how RIL affects surface soil properties and nutrients. We have attempted to put the observed changes in soils into the context of overall nutrient changes caused by logging across a patchwork of damage intensity. Although changes in the soils are significant and persistent, the area most affected and most significantly changed is a very small proportion of the total area logged when RIL techniques are used well. As a result, we suspect that the major nutrient losses, due to removal of bole wood and in some cases, leaching losses, are the most important factors to consider when evaluating the sustainability of RIL. Medium-to-low intensity reduced-impact logging with a sufficiently long cutting cycle might be a sustainable use of these forests. We need a better assessment of nutrients in bole wood and in rainfall inputs across the Amazon basin to assess where RIL might be sustainable and improve estimates of what harvesting intensities and cutting cycles can sustain forest nutrients in each area. Of course, any harvesting scenarios must also consider the regeneration rate and effects on the composition of the forest trees. The elevated concentrations of Ca and Mg in soils after logging and the persistence of this for 16 yr is an interesting phenomenon that we still do not fully understand. We propose that changes in soil geochemical processes are important in the immediate enhancement of soil Ca and Mg, and we suspect that these changes continue to play a role the persistence of this pattern, accepting that external inputs may also be important.

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