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Element loss on rain forest conversion in East Amazonia: comparison of balances of stores and fluxes

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Abstract. Nutrient loss from forest ecosystems by burning and leaching can be estimated by comparing nutrient stores before and after the disturbance, and by measuring the nutrient fluxes during disturbance. We applied both methods to a clear cutting experiment near Belem, Eastern Amazon, in a correct time series of 15 months duration. Nutrient flux measurements include wood export, loss to the atmosphere by burning, and leaching. The latter was based on water flux simulation with a soil water model, and on element analysis of soil water extracts (ceramic cup lysimeter). Two plots with 33 and 92 $\mathrm{t\,ha^{-1}}$ of residual biomass (left after export of wood >7 cm diameter) were compared with a forest control plot. Store and flux balances agreed well in case of Na, K, Mg, and S, and partially also for Ca and total P. Deviations seem to be caused by erratic but statistically not significant variations of the soil stores (mainly C and total N). Spacial variability of soil parameters demand very high sample replication, especially after disturbance. Flux measurements are better but more costly.

Introduction

Rain forest conversion implies a disturbance of the nutrient cycle and nutrient loss through biomass export, burning and leaching. This loss is unavoidable but a specification of the loss processes is important for a modification of the conversion methods. The quantification of the nutrient losses involved is often based on a comparison of nutrient outflow from differently managed water catchments, or of above- and below-ground element stores of plots representing systems before and after disturbance (false time series). A closer approximation should be possible by accompanying the actual process of disturbance on the same plot or in the same catchment in a correct time series. However, these studies cannot normally be extended over a longer time period. We tried to partly remedy this disadvantage by balancing both, fluxes and stores of elements, during a period of 15 months. This is also a comparison of two methods which are frequently applied but rarely ever to the same object.

For the estimation of fluxes with the soil water, the often used catchment approach has the disadvantage that it cannot differentiate between horizontal variations in site conditions and treatments (experimental plots) within the same basin, nor between processes in the rooting zone (mineralisation, uptake) and those in the deeper parts of the soil (weathering, exchange) (Bruijnzeel 1990; Brouwer 1996; Fölster and Khanna 1997; Stoorvogel et al. 1997). Because of these handicaps, and because of the physiographic characteristics of our study region – a flat topography without possibility of delineating distinct drainage basins – we determined the soil water flux by means of a soil water model (Klinge et al. 2001), and the element load transported with the soil water by ceramic cup extraction.

The study was part of a German–Brazilian cooperative research project (SHIFT, Study of Human Impact on Forests and Floodplains in the Tropics) on the sustainability of plantation forestry. We tried to combine the study of nutrient loss during rain forest conversion, and that during rotational plantation disturbance, by simulating the latter process in a natural forest in which two conversion plots with different residual biomass were established. The field study was carried out between 1992 and 1994 in Eastern Amazon (Klinge 1998).

Site and methods

Study site and experimental design

The experimental site was located at the field station of the Federal Cacao Research Institute, 17 km east of Belém-Pará, Brazil. Natural vegetation was a tropical lowland rain forest, partially exploited more than 40 years ago. Five-hundred-and-forty-two trees >10 cm DBH, belonging to 162 species, were registered on 750 m^2. With a basal area of 24 $m^2\,ha^{-1}$ and an above-ground biomass of 315 $Mg\,ha^{-1}$, this forest falls into the range of Amazonian rain forests (Mackensen et al. 2000). The soils are derived from late Tertiary sediments, and identified as deeply weathered Xanthic Ferralsols (UNESCO-ISRIC 1990). An organic surface layer exists but is weakly developed (14 $Mg\,ha^{-1}$).

The average annual precipitation of the area is about 3000 mm (EMBRAPA 1988; DENPASA 1995) and shows a seasonal pattern with high rainfall from January to April, moderate rains from May to September, a low rainfall season (October–November), and a transitional December. During the experimental period, rainfall was below normal. During the year September 1992 to August 1993, only 2479 mm precipitation was registered because of lower than average precipitation rates from August to December 1992 and in February 1993. The mean annual temperature is 26.4 °C, and the mean relative humidity 84% (EMBRAPA 1988).

We established three plots of 2500 m^2 each (see Figure 1) of which one plot (A1) remained under forest (control). Two others plots (A2 and A3) were clear-cut in September 1992. Stem biomass (>7 cm diameter) was removed manually with least disturbance. A residual biomass (leaves, twigs, branches <7 cm diameter, litter) of 33 $t\,ha^{-1}$ (A2) and 92 $t\,ha^{-1}$ (A3) remained behind to dry and was burned in November 1992. In February 1993, 3-month-old *Eucalyptus urophylla* seedlings were planted in both plots. The plots were frequently weeded until August 1993 but only

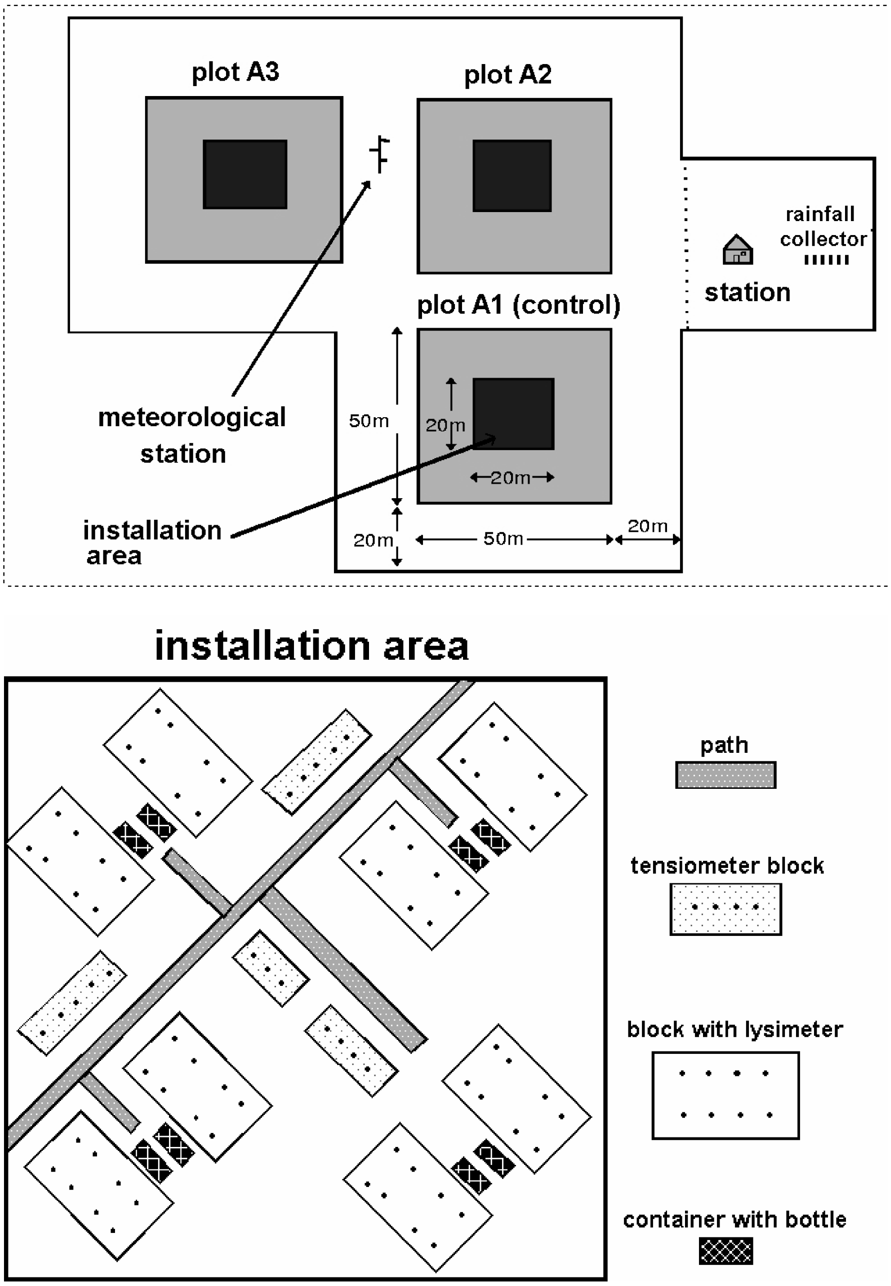

Figure 1. Plot lay-out and the arrangement of tensiometer and lysimeter blocks on each plot.

once afterwards. By November 1993, the eucalyptus reached a height of 2–3 m and a dry biomass of 0.5–0.8 Mg ha^{-1}.

Field and laboratory methods

The live and dead above-ground biomass was estimated by complete tree inventory (>7 cm stem diameter) and a regression analysis based on 21 sample trees, and by complete harvest of undergrowth and organic surface layer on 10% of the plot surface. The trees were divided in eight, the undergrowth in six compartments (for details see Mackensen et al. 2000), and analysed for chemical composition. The reported export of elements with the tree stems is based on the inventory.

Element loss due to volatilisation and particle transport was estimated by means of 72 brass plates (46 cm × 46 cm) placed below the dried residual biomass and the organic surface layer before burning. After the burn, the ash was collected and analysed. The residual biomass was calculated from the detailed estimate of the above-ground biomass components. Their element content was corrected by the amount of elements leached during the drying phase. Details are given in Mackensen et al. (1996).

Beginning in August 1992, before clear cutting started, we installed (see Figure 1):

- Meteorological equipment [open rainfall collectors, Thermo-Hygro-Sensor (Pt 100), 3-cup-anemometer (Thies), and a Pyradiometer (Thies) for separate global and reflex radiation] in a clear cut 50 m × 50 m area next to the experimental site, from February 1993 between A2 and A3 (Figure 1). In the forest control plot, throughfall (38 fixed collectors) and stem flow (12 adjacent trees with 400 m^2 total crown projection) were collected daily (for details see Klinge et al. 2001).
- Four tensiometer blocks (Figure 1) per plot with tensiometers at 10, 25, 40 cm (four each), 95 cm (three each), 145 and 500 cm (two each) depth. An automatically recording field station (laptop with switchboard) was used to record the meteorological and tensiometer readings every 15 min for the soil water model.
- Four blocks of suction-cup lysimeters (from February 1993 eight) (P 80, Staatliche Porzellan-Manufaktur, Berlin) with each two lysimeters per depth layer 25, 40, 60, and 110 cm, connected to the same collecting bottle, thus providing 4/8 samples of soil solution per plot, depth and date for analysis. The lysimeters had been washed in dilute HCl and distilled water, and afterwards conditioned with soil solution, and they were placed in auger holes at a distance which evades mutual influence in the same depth layer. Chloroform was added to each bottle to sterilise the solution during the collection period of 2 weeks. A suction between 0.4 and 0.8 bar was applied to individual bottles (by means of a manometer-controlled portable electric pump) depending on tensiometer readings.

Both, tensiometers and lysimeters were installed 10 cm below soil surface so that reading and collection of soil solution could continue before and throughout cutting and burning operations. The number of tensiometers and lysimeters was mainly based on previous experience on spacial heterogeneity of soil solution concentrations but was also a technical and financial compromise.

The soil water model (SilVlow, see Klinge et al. 2001) is based on the Richards equation (Ward and Robinson 1990). The model calculates one- or two-dimensional laminar water flow in porous media under transient saturated and/or unsaturated conditions. The distribution of water potential, soil water contents and water flow within and out of defined flow compartments are calculated with infiltration and evaporation as boundary conditions. The special feature of the model is that the initial soil parameter functions (volumetric moisture/matric potential (pF) and matric potential/hydraulic conductivity ($K(h)$) are introduced into the model in tabular form and can be modified by a fitting procedure using the matric potential measurements at six depths as a control for a backward calculation (Beven 1993). A high resolution measurement of pressure heads (15 min intervals) allows to follow the wetting front caused by individual rain events. Optimisation of the soil parameter functions were achieved during the vegetation-free period after cutting. For transpiration we used potential ET according to Penman (1948) corrected by a matric potential dependent transpiration reduction factor (Feddes et al. 1978) which was adapted to vertical root distribution and a fitting procedure between measured and simulated pressure heads. During the optimisation process, no need arose to consider bimodal porosity (preferential flow). The soils are not structured but contain only micro-aggregates (pseudosand).

On each plot, six undisturbed soil samples (8 cm diameter, 5 cm height: 250 cm^3) in each of five depth zones, were collected with a soil sampling kit (Eijkelkamp, The Netherlands) for the determination of volume weight and initial pF-curves. Samples for chemical analysis were collected by auger from 24 points per plot in the following depth layers: 0–5, 5–10, 10–20, 20–30, 30–50, 50–80, and 80–110 cm. They were mixed to give four analytical samples per depth layer. Sampling was carried out in August 1992, and repeated in November 1993.

All analysis was carried out in the soil laboratory of the Institute for Soil Science and Forest Nutrition, Göttingen, Germany. In soil solution, rain and throughfall, cations and total P (P_t) and S (S_t) were determined by ICP-AES, NH_4- and NO_3-N, total N (after UV treatment),and Cl colorimetrically. Only 8% of total N occurs as NH_4 (2–3%) and organic –NH_2).

In soil samples, cations were extracted with 1 N NH_4Cl and determined by AAS, total P colorimetrically after pressure digestion, total C and N by CN-Analyser. Plant and ash samples were analysed for total C and N by CN-analyser, and by ICP-AES for cations, total P and S after HNO_3 pressure digestion. All determinations were interspersed with about 20% repetitions, internal and external standards.

Nutrient fluxes in the soil were calculated by multiplying mean fortnightly element concentrations of the soil solution with the daily water fluxes obtained with the soil water model (Klinge et al. 2001).

Results and discussion

Changes of soil nutrient stores

The Ferralsols of the study area are dominated by medium (about 40%) and fine sand (about 25%) and contain about 20% clay with a calculated effective CEC of 3.1 $\mathrm{cmol_c\,kg^{-1}}$ clay (Mackensen et al. 2000). Under forest, total C drops from 18 $\mathrm{mg\,g^{-1}}$ (0–5 cm) to 3 $\mathrm{mg\,g^{-1}}$ (80–100 cm), the C/N-ratio from 15 to 12. Al is the dominant exchangeable cation with 70–80% saturation below 20 cm depth. Base saturation reaches 50% in 0–5 cm but drops to 20% below 20 cm. In the beginning of the experiment, the three plots did not show any significant difference in element concentrations (except total P). Therefore, soil element stores (before cutting) in Table 1 are based on the results from all plots. Stores calculated separately for each plot (see Table 1) differed (0–20%) with no uniform trend, due mainly to also non-uniform variations in the bulk densities per depth layer.

Table 2 shows the difference between element stores of the three plots before and after the experimental phase of 15 months. The differences cannot be evaluated statistically because the stores were calculated with mean concentrations and apparent densities. Significant differences of concentrations were only found in the 0–5 cm depth zone.

In the control area A1, the repeated analysis resulted in lower values throughout. Negative changes are also found in C and N of plot A2 and A3 in all depth zones. The % reduction is distinctly greater in total N than in total C on all three plots. As above-ground C and N are lost to the atmosphere to 94–98% during the fire (Mackensen et al. 1996), additions to the soil cannot be expected. An increase of 20–28% of total P in 0–10 cm depth of A2 and A3 can at least partly be explained by leaching from the residual biomass before burning (Mackensen et al. 1996). Higher total P values were found by Kauffman et al. (1995) and Tomkins et al. (1991) even 2 years after the disturbance, while Brinkmann and Nascimento (1973) registered this effect only for 5 months.

According to the scale of Fearnside (1990), the burn was excellent on plot A3 with the higher residual biomass, and moderate on A2. The known ash effect (Nye and Greenland 1964) on the pH is still apparent 12 months after the burn. The pH($CaCl_2$) of the upper 5 cm is increased by 1.6 units in A2, and by 1.8 units in A3. In 5–10 cm depth, an increase (0.5 units) was found only in A3.

Ca stores in the soil increased considerably on both converted plots, mainly in the 0–10 cm depth zone. This supports the observation of Brinkmann and Nascimento (1973) and Eden et al. (1991). In case of Mg and K, this effect is still visible in A3 though K has been leached and is concentrated in the soil below 30 cm. Na was presumably leached beyond 110 cm and shows no increase after 1 year. This sequence of relative mobility ($Na > K > Mg > Ca$) is confirmed by Nye and Greenland (1964), Brinkmann and Nascimento (1973), Sanches and Salinas (1981), and Tomkins et al. (1991). It also reflects the solubility trend of the respective carbonates in the ash (Raison et al. 1990).

Table 1. Element stores, apparent density, pH, and effective exchange capacity of the soils before the experiment (means of three plots) (AD: apparent density).

Depth (cm)	AD ($g\,cm^{-3}$)	pH_{CaCl_2}	C_t ($kg\,ha^{-1}$)	N_t ($kg\,ha^{-1}$)	P_t ($kg\,ha^{-1}$)	H ($kg\,ha^{-1}$)	Na ($kg\,ha^{-1}$)	K ($kg\,ha^{-1}$)	Ca ($kg\,ha^{-1}$)	Mg ($kg\,ha^{-1}$)	Fe ($kg\,ha^{-1}$)	CECe ($mmol_c\,kg^{-1}$)
0–5	1.49	3.5	13464	904	30.1	3.32	31.8	21.1	132.3	17.7	13.2	2.65
5–10	1.49	3.6	9129	644	27.8	1.51	29.7	13.2	38.7	8.1	12.6	2.19
10–20	1.53	3.8	14040	994	53.1	0.21	62.4	17.4	58.4	12.0	15.4	1.86
20–30	1.53	4.0	11556	790	47.9	0.00	61.5	15.0	43.3	9.8	8.5	1.68
30–50	1.53	4.1	19187	1405	116.3	0.00	120.7	20.8	61.0	14.8	10.5	1.53
50–80	1.59	4.1	21518	1634	155.8	0.00	188.1	29.0	85.6	20.4	11.6	1.31
80–100	1.59	4.1	6960	584	66.8	0.00	84.2	14.0	37.3	9.6	1.6	1.11
0–100			95855	6955	497.7	5.05	578.4	130.5	456.6	92.3	73.5	
A1			91200	6596	560		589	138	416	93	77	
A2			97477	6523	520		553	114	407	83	78	
A3			99013	7759	409		593	139	548	101	65	

Table 2. Difference between element stores before and at the end of the experiment, of plots A2, A3 and A1 (control), in kg ha^{-1} and in % of original stores (negative figures indicate apparent loss).

Plot	Depth	C_t (%)	N_t (%)	P_t (%)	K (%)	Ca (%)	Mg (%)	Fe (%)	Al (%)
A2	0–10	−6534 (−25)	−340 (−20)	9 (20)	−16 (−50)	215 (104)	7 (27)	−23 (−81)	−62 (−54)
	10–30	−4237 (−16)	−341 (−20)	−2 (−2)	−11 (−41)	−0 (−1)	−2 (−13)	8 (43)	16 (5)
	30–100	−3670 (−10)	−803 (−26)	−160 (−40)	−7 (−8)	45 (33)	−3 (−7)	24 (101)	182 (18)
	0–100	−14442 (−15)	−1483 (−23)	−153 (−29)	−34 (−30)	260 (64)	2 (3)	9 (11)	136 (10)
A3	0–10	−1398 (−5)	−214 (−12)	18 (28)	−10 (−19)	1085 (439)	38 (139)	−19 (−88)	−127 (−80)
	10–30	−1204 (−5)	−394 (−21)	−4 (−4)	11 (34)	80 (54)	10 (32)	4 (22)	−22 (−6)
	30–100	−4867 (−13)	−1244 (−33)	18 (4)	112 (194)	47 (30)	−4 (−7)	13 (43)	81 (7)
	0–100	−7470 (−8)	−1852 (−24)	32 (8)	113 (82)	1213 (221)	43 (43)	−3 (−5)	−68 (−4)
A1 control	0–10	−2458 (−13)	−179 (−13)	−18 (−27)	−6 (−22)	−35 (−26)	−6 (−22)	−8 (−30)	−0 (−1)
	10–30	−2589 (−11)	−222 (−14)	−22 (−20)	−10 (−32)	−26 (−28)	−4 (−22)	−5 (−21)	36 (11)
	30–100	−2306 (−6)	−667 (−19)	−168 (−44)	−32 (−44)	6 (4)	−8 (−17)	−0 (−8)	167 (21)
	0–100	−7353 (−8)	−1068 (−16)	−208 (−37)	−49 (−35)	−55 (−13)	−18 (−19)	−14 (−18)	202 (16)

Element concentrations in soil solution

The changing element concentrations in the soil solution during the experimental period from August 1992 to November 1993 are shown with the examples of the elements Ca (Figure 2) and K (Figure 3). There is a very prevalent peak in all elements from September to January. The concentration starts rising with the clear cutting of the forest and continues after burning. However, the peaks also reflects an element accumulation in the soil solution due to low rainfall in this period. With the onset of the rains in the end of January/February, dilution rapidly lowers the element concentrations. Rains during the dry period effectuate a vertical displacement with rising concentrations down to 60 cm, while a marked concentration increase at 110 cm depth follows the rains in February.

The Ca-concentrations (Figure 2) react strongly to clear cutting and burning. In 25 cm depth, the peak has subsided by March/April. The peaks in the soil depths of 40, 60, and 110 cm are retarded in time, and are successively lower. At least one smaller peak is recognised which follows a major disturbance by weeding. During the remaining time, the concentrations hardly exceed the ones in the forest control. NO_3^-, Mg, Cl, and Na concentrations show almost identical patterns except that the scales differ. NO_3 concentrations exceed that of Ca by a factor about 2, the Cl concentrations are in the same order of magnitude as Ca, while those of Na and Mg are smaller by a factor of 2 and 4, respectively. All of these elements react very strongly to clear cutting which may partly reflect the leaching from the aboveground residual phytomass (Mackensen et al. 1996) and the dead fine roots. However, the fast and strong reaction of NO_3 (93% of total N) shows that mineralisation and nitrification of the dead roots starts immediately after cutting (Montagnini and Buschbacher 1989; Steudler et al. 1991; Wong and Northcliff 1995; Palm et al. 1996). The reaction of Cl and Na proves their participation in uptake and release (Brouwer 1996). Analysis of selected tree components showed the presence of 1.5–3.5 mg g^{-1} Cl in tree leaves and up to 12.5 mg g^{-1} in epiphytes.

The pattern of K-concentrations in soil solution (Figure 3) is basically similar to that of Ca but the reaction is retarded. The maximum is reached only in January/February, except in A2 at 25 cm depth which is exceptional because of extreme values in four successive extracts of one lysimeter. After April 1993, the concentration falls in A2 to almost the level of the forest control. In A3, the concentrations fluctuate between 3 and 10 mg l^{-1} but without a vertical order of depth zones. This was also observed in case of Na. The retarded reaction of K is not yet well understood as K is normally leached very rapidly from organic debris. The pattern may possibly result from a complex interaction with Ca and the exchange complex of the soil.

The element concentrations show decreasing peaks with depth. This effect was also observed by Wong et al. (1990) and Sommer (2000) who discussed it under the aspect of plant uptake and/or adsorption by the soil. Uptake by plants is unlikely because of the sparse vegetation at this time. Adsorption by the soil is feasible. In order to arrive at a quantitative estimate of the amounts involved, we add here (Table 3) an outflow balance of elements at the sequence of depth zones. The flux

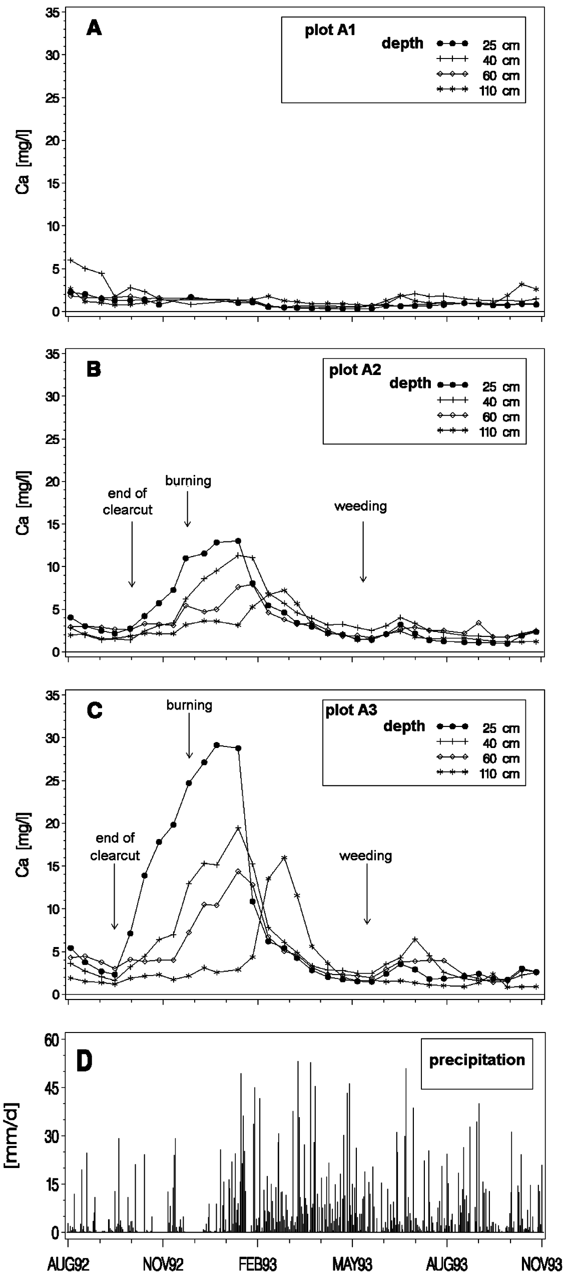

Figure 2. Fluctuations of rainfall (D) and Ca-concentrations in the soil solution at four depths of a forest control (A) and two conversion plots (B, C), during the experimental period from August 1992 to November 1993.

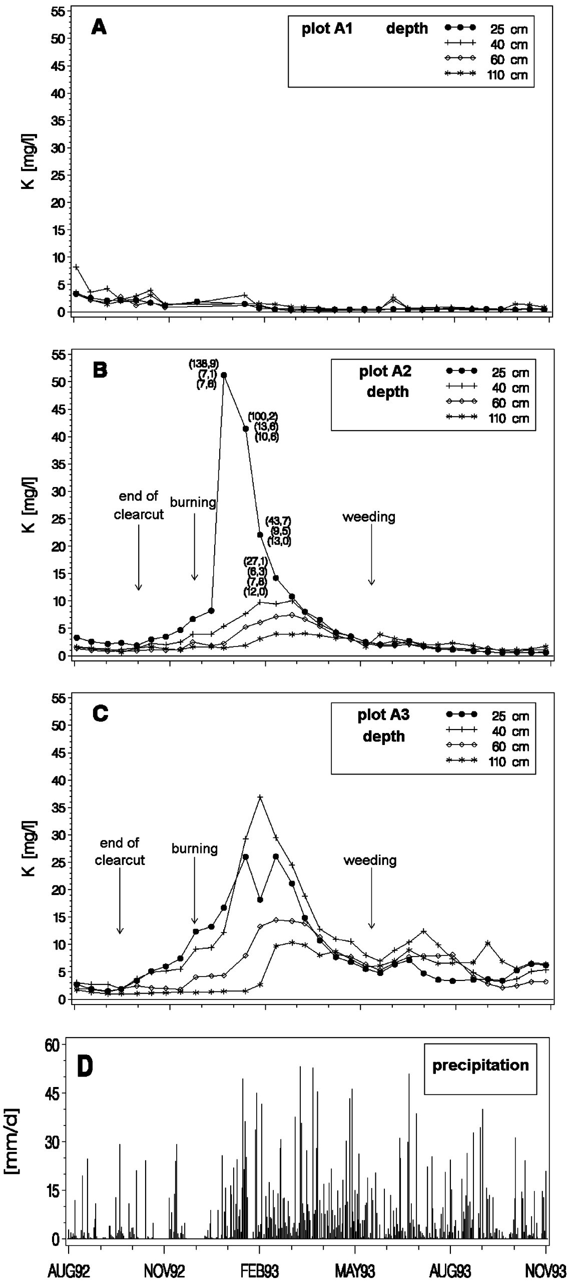

Figure 3. Fluctuations of rainfall (D) and K-concentrations in the soil solution at four depths of a forest control (A) and two conversion plots (B, C), during the experimental period from August 1992 to November 1993 (in brackets: variation of concentrations in individual lysimeters).

difference between 25 and 110 cm depth indicates the changes: under forest (A1), the flux looses water (as part of evapo-transpiration) but practically no elements. In the conversion plots, the situation is reverse. Most water runs through the soil column but considerable amounts of K, Ca, NO_3 and SO_4 are apparently absorbed. The small differences in Na and Cl can be interpreted as a result of their high mobility. About 9 (A2) and 15 (A3) $\mathrm{kmol_c\,ha^{-1}}$ K, Mg, and Ca are retained between 25 and 110 cm which is less than 10% of the effective cation exchange capacity of the soil column. The dominating exchangeable Al may be replaced, possibly precipitate with SO_4 or as gibbsite though there is also an efflux of 2–6 $\mathrm{kmol_c\,ha^{-1}}$. Less easily explained is the retention of 10 (A2) to 16 (A3) $\mathrm{kmol_c\,ha^{-1}}$ NO_3 and SO_4 as this corresponds to the total effective anion exchange capacity of the soil column (about 1 $\mathrm{mmol_c\,kg^{-1}}$, Anurugsa 1998), and there is no obvious efflux of replaced anions. Sommer (2000) found in similar soils after cut and burn of secondary bush fallow, also in Eastern Amazonia, that NO_3 and SO_4 which remained in the outflow below 1 m, were completely retained in the soil column between 1 and 3 m. The author took up a concept of Wada (1984) and Katou et al. (1996) suggesting that anion retention increases with the ionic strength of the solution after burning (or fertilisation) but decreasing with later dilution of the solution. He therefore uses the term retarded leaching. However, this problem remains to be solved.

Element balances of stores and fluxes

Balances of element stores and element fluxes for the 15 months period (1 September 1992 to 30 November 1993) are juxtaposed in Tables 4 and 5 for the plots A2 and A3, respectively. More wood was exported from A2. Element loss by wood export was therefore higher on A2 than on A3. This was compensated on A3 by higher element losses through burning and leaching in case of the strongly volatile N and S but not in case of K, Ca, and Mg. Above-ground element stores were naturally reduced to about 5–10%.

A comparison of the soil stores between 1992 and 1993, however, shows an ambivalent result: an increase is observed for Ca in A2, and for all base cations in A3 while amounts of C and total N (and of total P in case of A2) are distinctly lower in 1993. As the loss of N by leaching is small (170–210 kg) compared to the decrease of soil total N (1500–1800 kg), and as a substantial gaseous loss of N due to decomposition of organic matter is not to be expected, we can only resort to the known problem of spatial variability of soil parameters, respectively, the insufficient replications of samples. This is supported by the generally high variability of apparent losses between depth zones, and also by the forest control plot which also shows (Table 2) an apparent decrease not only of C and N but of all elements in 1993 compared to 1992 though the plot remained undisturbed.

The balances of stores and fluxes agree very well in case of Na, K, Mg, and S. Differences are found in total N throughout, and partially in total P (A2) and Ca (A3). The differences can be expressed in % of the original ecosystem stores: In

Table 3. Flux passage at 25, 40, 60 and 110 cm depth of soil water and elements during 15 months experimental phase, and difference between flux at 25 and 110 cm depth (Diff. 25–110 cm), in control plot (A1) and conversion plots (A2, A3).

	Water (mm)	Na (kg ha^{-1})	Cl (kg ha^{-1})	K (kg ha^{-1})	Ca (kg ha^{-1})	Mg (kg ha^{-1})	NO_3–N (kg ha^{-1}	S (kg ha^{-1})	Al (kg ha^{-1})
A1									
25 cm	2129	47	68	14	14	8	23	11	10
40 cm	2066	78	65	18	22	10	31	8	10
60 cm	1952	55	56	11	16	11	41	4	12
110 cm	1778	80	59	15	23	9	10	3	1
Diff. 25–110 cm	352	−33	9	−1	−9	−1	12	8	9
A2									
25 cm	2728	117	105	273	115	30	207	96	50
40 cm	2712	76	78	113	125	34	213	36	50
60 cm	2685	78	80	88	93	27	169	21	41
110 cm	2655	107	84	66	77	31	153	8	33
Diff. 25–110 cm	73	10	21	207	38	−1	54	89	17
A3									
25 cm	2728	113	174	288	198	42	344	98	84
40 cm	2712	144	161	360	163	53	308	127	74
60 cm	2685	105	129	203	134	34	212	50	42
110 cm	2655	166	149	166	102	29	195	20	31
Diff. 25–110 cm	73	−54	24	122	96	12	149	77	53

Table 4. Balance of element stores and element fluxes during conversion of rain forest on plot A2 (1992/1993) (live biomass).

	Biomass (Mg ha^{-1})	C_t (kg ha^{-1})	N_t (kg ha^{-1})	P_t (kg ha^{-1})	Na (kg ha^{-1})	K (kg ha^{-1})	Ca (kg ha^{-1})	Mg (kg ha^{-1})	S_t (kg ha^{-1})
Soil element store 1992		98	6523	520	553	114	407	83	
Aboveground element store 1992	313	156	1741	55	134	797	1727	212	248
Total element store 1992		253	8264	575	687	911	2134	295	248
Soil element store 1993		83	5040	367	455	80	667	85	
Aboveground element store 1993	9	4	137	5	3	60	106	19	14
Total element store 1993		87	5176	372	458	140	773	104	14
Store balance 1992/1993		−166	−3087	−202	−229	−772	−1361	−191	−234
Element loss by wood export	279	140	1397	46	116	688	1397	175	206
Element loss by fire		15	325	2	3	7	44	7	26
Element loss by leaching			166	2	107	66	77	31	8
Total flux loss by conversion		155	1889	49	226	761	1517	212	240
Element input by rain			8	3	26	4	9	2	6
Flux balance		−155	−1880	−46	−200	−756	−1508	−210	−234
Leaching loss in % of total flux loss			5	4	47	9	5	15	3

Table 5. Balance of element stores and element fluxes during conversion of rain forest on plot A3 (1992/1993) (live biomass).

	Biomass ($Mg\,ha^{-1}$)	C_t ($kg\,ha^{-1}$)	N_t ($kg\,ha^{-1}$)	P_t ($kg\,ha^{-1}$)	Na ($kg\,ha^{-1}$)	K ($kg\,ha^{-1}$)	Ca ($kg\,ha^{-1}$)	Mg	S_t ($kg\,ha^{-1}$)
Soil element store 1992		99	7759	409	593	139	548	101	
Aboveground element store 1992	317	158	1681	51	124	722	1560	196	243
Total element store 1992		257	9441	461	717	861	2108	298	243
Soil element store 1993		91	5907	441	446	252	1761	144	
Aboveground element store 1993	10	4	169	8	3	91	139	25	17
Total element store 1993		96	6076	449	449	343	1900	170	17
Store balance 1992/1993		−161	−3364	−11	−268	−518	−208	−128	−225
Element loss by wood export	223	112	828	22	65	330	712	96	160
Element loss by fire		42	804	7	10	87	189	40	70
Element loss by leaching			211	2	166	165	102	29	20
Total flux loss by conversion		154	1843	31	241	583	1004	165	220
Element input by rain			8	3	26	4	9	2	6
Flux balance		−154	−1835	−28	−215	−579	−995	−162	−214
Leaching loss in % of total flux loss			11	6	69	28	10	18	9

Table 6. Comparison of element store in the residual phytomass and element loss by burning and leaching, on plot A2 and A3.

	C_t (kg ha^{-1})	N_t (kg ha^{-1})	P_t (kg ha^{-1})	Na (kg ha^{-1})	K (kg ha^{-1})	Ca (kg ha^{-1})	Mg (kg ha^{-1})	S_t (kg ha^{-1})
A2								
Element store in residue	16	343	9	18	109	330	37	42
Element loss by burning and leaching	15	491	3	110	73	120	37	34
Loss in % of residual store	94	143	38	595	67	36	100	81
A3								
Element store in residue	46	854	29	59	392	848	100	113
Element loss by burning and leaching	42	1015	8	176	253	292	69	90
Loss in % of residual store	91	119	29	298	65	34	69	80

A2, they are 15% (total N) and 27% (total P) but below 10% for the remaining elements; in A3, the respective figures are 16% (total N), 37% (Ca), and <12% (remaining elements). One could assume that agreement is generally better in elements which are mainly stored in the (original) forest vegetation than in elements preferentially stored in the soil. We tend to explain this trend with a greater spacial heterogeneity in soil parameters compared to soil solution chemistry. This conclusion favours the flux balance approach though it is certainly more costly.

Comparison of the result to that of other authors is handicapped by different site conditions and methodologies. Most studies base their estimates on a comparison of element stores of neighbouring plots before and after conversion (false time series). On a comparable Amazon site, Russel (1983) arrived at a % total loss of base cations (74–83%) similar as on our plot A2, Fölster et al. (1976) on sediments of the Colombian Magdalena Valley at 50–60%, more similar to our plot A3. Hase and Fölster (1983) and Bruijnzeel and Wiersum (1985) operated on fertile soils (young alluvial and volcanic soils respectively). As Hase and Fölster (1983) pointed out, the greater the element stores in the soil, the less likely it is to statistically verify differences in the before/after scenario. Our study shows that this problem arises even with smaller soil stores, and though we operated in a correct time series avoiding the uncertainty of the pre-conversion status of plots (Fölster and Khanna 1997).

Flux measurements were applied in a correct time series in Sabah, Malaysia, though with a catchment approach (Malmer and Grip 1994; Malmer 1996). The outflow after conversion showed comparable loss rates of K but much lower rates of other elements. However, the comparison is handicapped, as the authors considered the total catchment while we concentrated on processes and changes above and within the rooting zone.

Brouwer (1996) applying a similar lysimeter approach (in Guyana) as we in an experimental plot arrived at a similar order of magnitude of leaching loss during 2 years (without burning) as we on plot A2 in case of total N, Mg, and K but at a distinctly lower loss of Ca (30%). Amount and composition of residual phytomass is unknown. Leaching loss in a comparable experiment by Parker (1985) in Costa Rica equalled that of Brouwer (1996) but was lower in total N and especially K.

Of the two fluxes burning and leaching, the former contributes 2–4 times more in case of the volatile N and S. During the moderate burn of A2, less base cations were lost to the atmosphere than by leaching. Leaching of K still prevailed over volatilisation in the hot fire of A3 while this ratio reversed in case of Ca and Mg, mainly due to the greater particle transport in the hot burn (Mackensen et al. 1996). In general, it is likely that our plots which were surrounded by forest, experienced a reduced loss of ash particles and a relatively higher subsequent dissolution and leaching of ash. In open fields of small farmers agriculture in the region, Hölscher et al. (1997), (see also Mackensen et al. 1996) and Sommer (2000) found in their experimental burns of secondary forest fallows a higher loss to the atmosphere, and consequently a relatively lower leaching loss.

An aspect which is of special importance for plantation forestry, is summarised in Table 6. It shows the elements contained in the residual phytomass after clear cutting and the removal of stems, and the amount lost by burning and leaching. The loss expressed as % of the residual store indicates to what extent the released elements can be stored in the soil (Ca 65%, K 35%). High losses reflect strong volatilisation during the burn (C, S), additional decomposition of below-ground biomass (total N) and/or a high leaching rate (Na). When applying the results to plantation rotations, one should keep in mind that quantity of root mass is probably smaller, and less NO_3 will be released. Ruhiyat (1989) had suggested that the nutrient content of the residual phytomass might correspond to the loss by burning and leaching. This assumption is applicable in the sense that the loss is related to the nutrient store in the residual phytomass but has to be corrected by an element-specific factor. For the plantation manager this may be helpful when calculating the total nutrient loss (export and conversion loss) in plantation forestry.

Conclusions

The loss of nutrients during ecosystem disturbance can be estimated by measurements of fluxes or by comparison of nutrient stores before and after disturbance. In general, results of both methods agree well in elements mainly stored in the vegetation. In elements mainly stored in the soil (especially C and total N), comparison of stores provides a less satisfactory basis for estimating the loss, even in a correct time series. Flux measurements demand a good estimate of water fluxes and are costly. But the necessary increase of sample size in a store balance could partially compensate these costs.

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