Soil chemical changes after tropical forest disturbance and conversion: The hydrological perspective

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Abstract

The soil chemical and hydrochemical changes accompanying tropical forest disturbances of varying intensity are reviewed. Various methodological problems are discussed with respect to the establishment of ecosystem nutrient balances and the determination of changes therein due to forest exploitation or conversion to other land uses. It is shown that amounts of nutrients removed from the forest ecosystem in harvested timber generally exceed those associated with enhanced leaching, both after selective logging and clearcutting operations. Losses of base cations through volatilization and ash dispersal upon burning residual biomass are substantial, contrary to common belief. Approximate periods required for the replenishment of lost nutrients at the ecosystem level are estimated by comparing the losses with inputs via bulk precipitation and/or mineral weathering. The results suggest typical recovery periods of 30–60 years, depending on harvesting intensity, rainfall regime and soil nutrient retention capacity. Issues requiring further research include: (i) the evapotranspiration from young regenerating and planted vegetation; (ii) nutrient losses via rapid flow through macropores during rainfall (as opposed to slow matrix flow); (iii) nutrient losses via erosion after mechanized tree harvesting and burning of slash (particularly in the context of short-rotation plantation forestry); and (iv) the possible contributions to the overall ecosystem nutrient budget by mineral weathering. Particular emphasis is placed on the integration of hydrological, pedological and plant ecological process research at various levels of scale within an overall catchment ecosystem context in order to facilitate the integration of different disciplines. A plea is made to concentrate future research efforts at a relatively small number of carefully selected and well-researched locations, possibly joined together in a network capturing the chief environmental variability encountered in the humid tropics.

1 Introduction

The classical view of a tropical rain forest as having a relatively rich internal nutrient economy perched on a nutrient-poor substrate has been shown by *Proctor* (1987) to be an overgeneralization. In addition, the areal extent of extremely infertile soils in the tropics appears to be much smaller than previously assumed (*Richter* and *Babbar*, 1991). Nevertheless, the notion that tropical forest disturbance or conversion to other land-use types will invariably lead to environmental havoc (including dramatically reduced soil fertility) – aptly summarized by the phrase 'from green hell to red desert' – is still widespread. As suggested by *Hamilton* and *King* (1983) and *Jordan* (1985) it is important in this respect to recognize the intensity of a specific disturbance rather than lump everything under such meaningless terms as 'deforestation'.

The hydrological and soil physical impacts of various kinds of tropical forest disturbance and conversion have been discussed in detail in a series of earlier review papers by the author (*Bruijnzeel*, 1990, 1992a, 1993, 1996, 1997). The present contribution focuses on the accompanying changes in soil chemical characteristics. In doing so, a specific attempt will be made to highlight the close link between hydrology and soil science in this respect. The paper starts with a brief discussion of the solute losses from undisturbed (lowland) tropical rain forest, against which any changes due to forest exploitation or clearance can be compared. Next, the respective soils impacts of various degrees of forest disturbance are reviewed in order of increasing intensity. After identifying a number of issues requiring further research, the paper concludes with the suggestion that future work should preferably be integrating hydrological and biogeochemical aspects at various levels of scale within an overall catchment ecosystem context.

2 Solute losses from undisturbed lowland forests

Tropical forests growing on very nutrient-poor substrates are only able to maintain a high biomass level through an array of nutrient conserving mechanisms, together producing a relatively 'tight' or 'closed' nutrient cycle with only small amounts of nutrients leaking from the system (*Herrera* et al., 1978; *Brinkmann*, 1985). Conversely, forests on more fertile substrates will exhibit a more 'open' type of nutrient cycle (*Baillie*, 1989). It is important to make a distinction here between primary and old secondary forests, which have essentially attained a state of dynamic equilibrium (*Uhl*, 1982; *Lieberman* et al., 1985), and young secondary forests or plantations which generally represent rapidly aggrading ecosystems. In the former the rate of nutrient uptake to maintain overall forest biomass will be covered by and large by the amounts of nutrients released from decomposing litter and logs, supplemented by contributions via atmospheric deposition (*Vitousek* and *Sanford*, 1986; Proctor, 1987). Net uptake rates in young secondary or plantation forests, however, normally exceed the amounts of nutrients cycled via litterfall and crown wash, especially during the first decade (*Ewel*, 1976; *Bruijnzeel*, 1983; *Jaffré*, 1985; *Gholz* et al., 1985). The remainder will have to be supplied by the soil and this may present difficulties under certain conditions. Examples include subsequent rotations of fast-growing exotics planted on very infertile soils (*Russell*, 1983; *Spangenberg* et al., 1996) or the repeated harvesting of nutrient-demanding hardwood species on fertile substrates (*Hase* and *Fölster*, 1983; *Bruijnzeel* and *Wiersum*, 1985). Although in the short run, nutrient availability is governed by the balance between processes releasing nutrients into available forms and those removing them (*Proctor*, 1987), in the long run the nutrient status of a forest ecosystem depends on the balance between overall nutrient inputs (atmospheric deposition, mineral weathering, gas absorption/fixation) and outputs (mainly leaching and volatilization). Because the ratio of solid matter (soil and weathered rock) over the liquid phase (percolating water) is very large, small changes in the chemical composition of the former may show up as quite detectable changes in the chemistry of the latter (*Verstraten*, 1980; *Bruijnzeel*, 1983). As such, amounts of nutrients carried in solution by low-order streams can be regarded as a useful, albeit relative, indicator of an area's nutrient status, both before and after a change in land use (*Likens* et al., 1977).

Bruijnzeel (1991) examined the results of 20 nutrient input-output budget studies for tropical forest ecosystems. Scatter plots of annual streamflow totals *vs.* annual calcium, magnesium and potassium losses revealed four groups with characteristic nutrient export patterns that corresponded closely with soil fertility levels. The results are shown in Figure 1 together with those obtained more recently for eight catchments in Malaysia (*Grip* et al., 1994; *Zulkifli* and *Abdul Rahim*, 1994), Central Amazonia (*Lesack*, 1993ab), Ivory Coast (*Stoorvogel*, 1993), Fiji (*Waterloo*, 1994), and Guyana (*Brouwer*, 1996).

As shown in Figure 1, calcium and magnesium losses from areas with spodosols and highly leached oxisols are distinctly lower than for the overall oxisol/ultisol group. In turn, losses from sites with inceptisols exceed those from the latter group whereas still higher exports have been reported for forested catchments underlain by rocks that are particularly rich in calcium or magnesium, such as limestones, calcareous shales and slates, or ultramafic rocks (see *Bruijnzeel* (1991) for details). Potassium exports exhibit a similarly upward trend over the first three groups, with some overlap between groups. Potassium losses from the fourth group (soils of high fertility) are low, reflecting the scarcity of the element in the corresponding bedrock types (Figure 1c).

Interestingly, exports of base cations in streamflow from areas with highly infertile soils (and presumably tight nutrient cycles) were nevertheless quite high for streams draining catchments larger than, say, 10 km^2. Solute losses from such areas were much smaller when evaluated for small (several hectares) catchments and/or plots (*Bruijnzeel*, 1991). Such discrepancies may be interpreted in terms of the relative depths of (i) the weathering front (where solutes are released), (ii) the degree of river incision into the substrate, and (iii) the fine root network (*Baillie*, 1989; *Burnham*, 1989; *Bruijnzeel*, 1991; *Eernisse*, 1993). This finding not only has implications for the experimental evaluation of solute losses from tropical forest ecosystems but also for the overall availability of nutrients to the vegetation. The role of deep roots in tropical rain forests has received insufficient attention (cf. *Eernisse*, 1993; *Nepstad* et al., 1994).

3 Chemical responses to forest disturbance

Site fertility is potentially threatened upon forest exploitation or conversion by the removal of nutrients in harvested timber, through surface erosion, and via enhanced leaching as a result of (i) the larger volumes of water percolating through the soil after opening up of the canopy; (ii) a temporarily reduced capacity of damaged vegetation to take up nutrients; and (iii) the sudden addition of large amounts of fresh organic debris to the forest floor that are either left to decompose or burned to facilitate future access for planting (*Martin*, 1970; *Ewel* et al., 1981; *Uhl* and *Jordan*, 1984; *Poels*, 1987; *Bruijnzeel*, 1990; *Sim* and *Nykvist*, 1991). Naturally, infertile sites will be particularly vulnerable because much of the ecosystem's nutrient capital will be stored in the vegetation rather than in the soil and the disturbance may bring about the partial or complete disruption of the chief nutrient conservation mechanisms (especially the surface root mat; *Jordan*, 1985). In the following we will examine the hydrochemical and soil chemical responses to disturbances of (i) low to intermediate intensity (treefalls, hurricanes, logging, forest fires, and shifting cultivation); and (ii) high intensity (mechanized clearfelling followed by burning and conversion to other land uses, primarily forest plantations).

Basically, two approaches have been followed when trying to evaluate such responses, viz. the study of changes in (i) (top)soil nutrient concentrations over time (e.g. *Gillman* et al., 1985; *Amir* et al., 1990), and (ii) the chemical composition of the drainage water or streamflow (e.g. *Parker*, 1985; *Zulkifli*, 1989; *Malmer*, 1993). Each method has its specific difficulties and limitations. The soils approach suffers primarily from problems related to

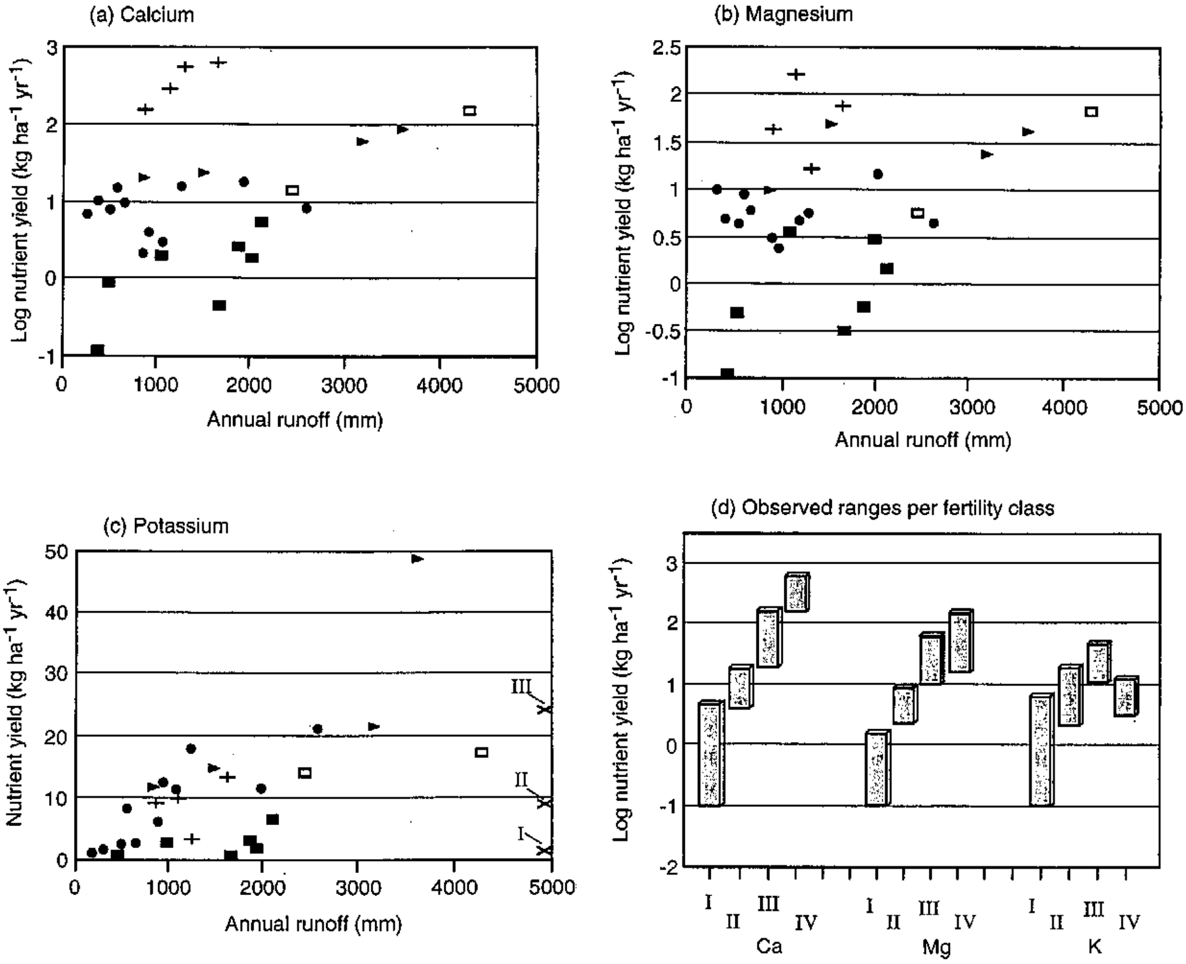

Figure 1: Solute losses of (a) calcium, (b) magnesium and (c) potassium (kg ha^{-1} yr^{-1}) vs. amounts of streamflow and the ranges in output per soil fertility class (I–IV) for 28 tropical forest ecosystems (updated from Bruijnzeel, 1991). Legenda: ■ Group I, soils of very low fertility (spodosols, oxisols); ● Group II, moderately infertile soils (ultisols, some oxisols); ▶ Group III, moderately fertile soils (inceptisols); and + Group IV, soils of high fertility (mollisols, vertic soils); □ large catchment; × group average (potassium only, Figure 1c).

high spatial and/or temporal variability, particularly in such diverse ecosystems as tropical rain forests (*Proctor* et al., 1983; *Burghouts*, 1993) but also in even-aged plantations (*Lundgren*, 1978; *Hase* and *Fölster*, 1983). In addition, when dealing with long-term changes in soil nutrient reserves (e.g. during the respective phases of the shifting cultivation cycle or a rotation of plantation forest), the sampling of a single site repeatedly through time becomes often impractical and one will have to resort to sampling a 'false time series' representing the respective stages of vegetation development or the number of previous rotations. Needless to say, high spatial variability may easily confound the results obtained with this method (cf. *Hase* and *Fölster*, 1983; *Buschbacher*, 1984; *Spangenberg* et al., 1996).

Bruijnzeel (1991) discussed the numerous pitfalls associated with the hydrological approach at various levels of scale, whereas *Sollins* and *Radulovich* (1988) and *Lesack* (1993a) stressed the importance of the proper quantification of nutrient losses in macropore flow c.q. stormflow (cf. *Russell* and *Ewel*, 1985). Indeed, reliable estimates of nutrient losses via leaching are difficult to obtain. Within the present context, various investigators have used zero-tension or suction lysimetry to study the chemical composition of free-draining and matrix soil water, respectively, after experimentally cutting or burning tropical forest (e.g. *Toky* and *Ramakrishnan*, 1981; *Uhl* et al., 1982; *Russell*, 1983; *Brouwer, 1996 & this volume*). In the absence of sound hydrological measurements, however, the corresponding solute losses obtained by several of these studies must be considered doubtful (*Jordan*, 1989; *Bruijnzeel*, 1990, 1991). There is a dearth of reliable information on the water use of young regenerating vegetation in the humid tropics (*Hölscher*, 1995; *Roberts* et al., 1996).

Others have attempted to avoid the problems associated with the quantification of the rapid component of soil drainage (*Cooper*, 1979; *Russell* and *Ewel*, 1985) by evaluating nutrient losses after forest clearing and burning at the small catchment scale (*Malmer* and *Grip*, 1994; *Waterloo*, 1994). Although the amount of streamflow leaving the catchment may be established with sufficient accuracy (*Lee*, 1970), its chemical composition may differ from that of the soil water taken up by the vegetation on the sideslopes (*Nortcliff* and *Thornes*, 1978; *Bruijnzeel*, 1983). Type, fertility and degree of aeration of soils in the riparian zone may all be dramatically different compared to conditions found on the slopes and often result in a distinctly different vegetation type as well (*Brünig* et al., 1978; *Proctor* et al., 1983; *Poels*, 1987; *Johnston*, 1992). As such, a combination of plot- and catchment-based methods is likely to give the best results (*Bruijnzeel*, 1991; *Elsenbeer* et al., 1994; *Grip*, 1994).

As for the quantification of amounts of nutrients carried away in eroded sediment, a catchment-based approach will generally be less suitable because the resulting sediment yields will contain contributions from various sources other than the hillslopes themselves (stream bank and bed erosion; gullies; landslides, etc.). In addition, there is the problem of (temporary) storage of already eroded material in various positions in the landscape which tends to obscure on-site losses (*Walling*, 1983; *Rose*, 1993). Generally, bounded runoff plots have been used to measure surface erosion but these too have their disadvantages (*Lal*, 1988): *Wiersum* (1984) reviewed the results of more than 80 erosion studies in various tropical forest and tree crop systems. His conclusions were that (i) low erosion rates (< 1 t ha^{-1} yr^{-1} on average) prevailed in undisturbed and regenerating natural forests, multistoried tree gardens, and in forest plantations and tree crops with either a well-developed litter layer or some form of cover crop/mulch; (ii) intermediate levels of erosion (2–10 t ha^{-1} yr^{-1} on average; range 0.4–70) were observed during the cropping phase of the shifting cultivation cycle or during the intercropping stage of plantation forest establishment; and (iii) truly high erosion rates (c. 50 t ha^{-1} yr^{-1} on average; range 1–183) occurred in the case of clean-weeded tree crops or forests where the protective litter layer was destroyed by fire or harvesting. These conclusions have been confirmed by later work (*Fritsch* and *Sarrailh*, 1986; *Young*, 1989; *Nortcliff* et al., 1990; *Malmer*, 1993; *Ross* and *Dykes*, 1996). The nutrient content of either the surface runoff or the eroded material was not discussed by *Wiersum* (1984). Indeed, the associated nutrient losses have usually been quantified within an agricultural rather than a forest context (*Stocking*, 1984), although there are exceptions (e.g. *Kang* and *Lal*, 1981; *Roose*, 1981; *Hudson* et al., 1983; *Lima*, 1988; *Ross* et al., 1990; *Ramakrishnan*, 1992). The results of these studies are rather variable (if not contradictory at times), however, and no clear patterns emerge other than the obvious conclusion that more erosion implies more nutrient loss. Also, eroded sediment does not necessarily remove more nutrients from the site than does the runoff water. There is a need for more work on this aspect, particularly with respect to the nutrient losses in eroded material associated with mechanized forest harvesting and burning large volumes of slash (cf. *Malmer*, 1993, 1996).

3.1 Response to disturbances of low to intermediate intensity

Treefalls, logging and hurricanes

Studies of spatial variations in topsoil nutrient concentrations in mature-phase forest and small (47–164 m^2) *treefall gaps* in Costa Rica and Venezuela (on volcanic and old sedimentary substrates, respectively) were unable to detect any systematic differences that could be ascribed to the addition of organic matter in places where the bole and the crown of the fallen tree were downed (*Vitousek* and *Denslow*, 1986; *Uhl* et al., 1988b). Similarly, no significant increases in the nutrient concentrations of free-draining soil water were recorded for such gaps (*Parker*, 1985; *Uhl* et al., 1988b). However, a distinct rise in solute concentrations in soil moisture at 70 cm depth has been reported to occur about four months after the creation of artificial gaps of 500 m^2 (10 m x 50 m) and 2500 m^2 (50 m x 50 m) in lowland rain forest on an andept in Costa Rica (*Parker*, 1985). The pulse of elevated concentrations lasted for about a year and reached a peak some 6–8 months after gap creation (Figure 2a). Interestingly, concentrations started to rise earlier in the larger gap, which also experienced the highest soil moisture levels and, therefore, hydraulic conductivities (Figure 2b). Concentrations of nitrate showed a three- to fourfold increase while calcium and magnesium concentrations roughly doubled. Neither potassium, nor phosphate or ammonium concentrations responded markedly after clearing, possibly because of vigorous uptake by pioneer vegetation and nitrification of the ammonium (*Parker*, 1985; cf. *Robertson*, 1984). Apparently, enhanced leaching losses may occur if the size of the gap reaches a certain threshold value which is possibly located somewhere between 200 and 500 m^2 (*Parker*, 1985; *Uhl* et al., 1988b). Estimated leaching losses of base cations during the first 13 months after cutting were such that they could be replenished by nutrient inputs via rainfall within a few years (Table 1).

No material was removed from the site in the above studies nor was there any disturbance of the soil surface as would be the case with *mechanized removal of timber* (*Bruijnzeel*, 1992a). *Uhl* et al. (1982) presented an interesting comparison in this regard for an area underlain by waterlogged spodosols in the Amazon Territory of southern Venezuela. A 50 m x 100 m forest plot was (manually) cut and left to regenerate whilst a nearby patch of 4 ha of forest was cleared by bulldozing. The effects of the two disturbances on vegetation structure and composition as well as the chemical composition of the topsoil were evaluated after three years. Despite significantly increased nutrient concentrations in free-draining soil water during the first two-and-a-half years after forest cutting the nutrient concentrations in the top 10 cm of the soil were significantly higher than those in the original forest soil. The intensity of leaching was greatest for potassium, magnesium and nitrate but after three years concentrations in the leachate had started to approach the levels observed under undisturbed forest again. Unfortunately, the corresponding amounts of leached nutrients were not given. Biomass accretion in the manually cut site was mainly by sprouting and had attained the rather low value of 1290 g m^{-2} after three years (*Uhl* et al., 1982). Leachate concentrations were not monitored at the bulldozed site but nutrient concentrations in the topsoil (and biomass) were much lower than before disturbance, mainly as a result of topsoil removal. Forest regeneration at this severely disrupted site was also slow, with only 77 g m^{-2} of biomass having accumulated after three years. Interestingly, woody regeneration was rapid around the margins of the bulldozed plot where topsoil and organic debris were piled up (*Uhl* et al., 1982). Another effect of the drastic treatment was the gradual dying of all the trees in the surrounding forest downslope of the bulldozed plot, presumably due to a rise in the groundwater table as a result of the greatly diminished water uptake by the vegetation in the bulldozed area (*Uhl* et al., 1982).

Gillman et al. (1985) investigated the impact of mechanized logging on soil nutrient concentrations in upland Queensland. The harvesting of 33% of the basal area associated with trees of 20 cm dbh and larger caused an initial reduction in canopy cover of 18%, which recovered to 91% of the original value within four years after logging. Statistically significant decreases in concentrations of organic

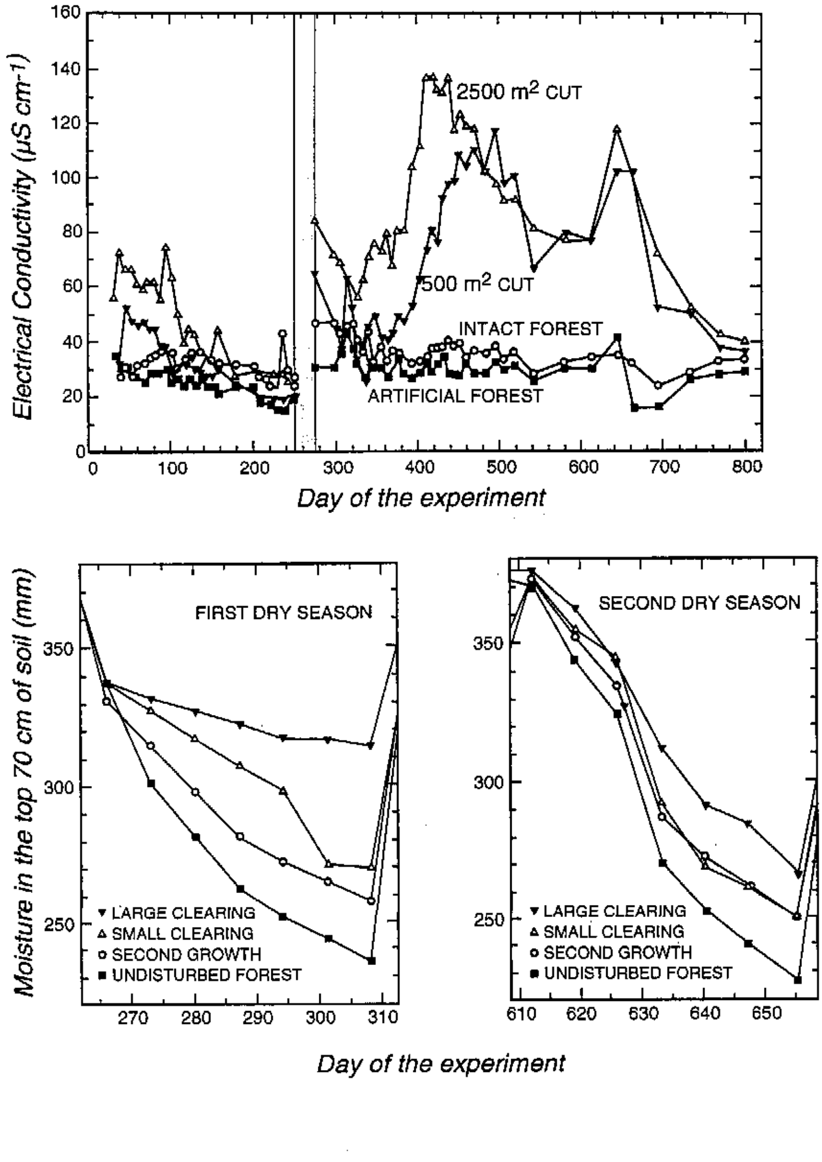

Figure 2: (a) Changes in the electrical conductivity and (b) amount of soil water at 70 cm depth in undisturbed forest, small artificial gaps and two clearings of contrasting size, Costa Rica (after Parker, 1985).

Table 1: (a) Nutrient losses in harvested timber and (b) increased leaching (kg ha^{-1}) after gap creation in tropical forest and the approximate time span (yr) required to compensate the losses via nutrient inputs from (c) atmospheric sources, and (d) *idem* plus weathering. Values above 10 have been rounded off to the nearest 5.

Location	Volume of extracted timber (m^3 ha^{-1})		Ca	Mg	K	Period for which leaching loss has been computed (yr)
			(kg ha^{-1})			
La Selva,	–	(b)	15	15	2	1.1
Costa Rica[1, 2]		(c)	*5*	*5*	*1*	
Mabura Hill,	ca. 200++	(a)	75	30	75	
Guyana[3, 4]	large gap	(b)	25	30	40	2.8
	small gap		20	15	15	
	large gap	(c)	*35*	*50*	*35*	
	small gap		*35*	*35*	*30*	
Bukit Berembun,	60	(a)	200	45	20	
Malaysia[5, 6*]		(b)	30	15	75	5
		(c)	*60*	*60*	*20*	
Kabo, Suriname[6, 7*]	20	(a)	80	7	35	
		(b)	3	2	2	2
		(c)	*30*	*5*	*10*	
Sipitang, Malaysia[8, 9*]	145	(a)	140	40	70	
		(b)	25	8	105	2.8
		(c)	*55*	*45*	*35*	
		(d)#	*10*	*4*	*20*	
Windsor Tableland,	80	(a)	95	20	10	
Northern Australia[10]		(c)	*35*	*20*	*5*	

[1]Parker (1985) (solute concentrations in soil water; nutrient input via precipitation);
[2]Bruijnzeel (1990) (water balance; his Table 4);
[3; 4]Brouwer (1996), *this volume;*
[5]Zulkifli (1989);
[6]Bruijnzeel (1992a);
[7]Poels (1987);
[8]Malmer (1993);
[9]Sim and Nykvist (1991);
[10]Gillman et al. (1985);
*basin study;
++65 and 16 m^3 of timber removed from the large and small gap, respectively (L. C. Brouwer, personal communication);
#estimated via nutrient outputs from undisturbed catchments with ultisols and spodosols (Malmer 1993), combined with proportions of catchment underlain by the two soil types (Sim and Nykvist, 1991); nutrient inputs via bulk precipitation according to Burghouts (1993).

matter, total nitrogen, pH, and exchangeable bases were found in skidder track depressions (occupying 13% of the total sample area of 5.6 ha) where topsoil had been removed by bulldozing (cf. *Uhl* et al., 1982). Similarly, increases were found for the mounds formed at the sides of the skidder tracks (15%), but concentrations were not significantly different in areas for which the only disturbance consisted of the addition of crown debris (40% of the area) or increased irradiation (16% of the area). On the basis of their well-planned stratified sampling programme, *Gillman* et al. (1985) concluded that, for the whole site, contents of nitrogen and exchangeable bases had not altered significantly after four years, although organic carbon had declined by about 15%. It would seem, therefore, that the only serious nutrient losses from this particular site would be those associated with stemwood removal and that losses via increased leaching were minor. Based on nutrient concentrations in bulk precipitation in the area (*Brasell* and *Gilmour*, 1980), it would take anywhere between four (potassium) and 34 (calcium) years before these losses would be compensated by atmospheric inputs alone (cf. Table 1). Therefore, the rotation period applied in the area (40–50 years) is probably sufficiently long for the ecosystem to recover from logging, suggesting that the Queensland polycyclic silvicultural system is sustainable from the nutrient point of view in this particular area.

Other studies of changes in soil chemistry over periods of one to two years after logging tropical forest include those by *Enright* (1978) in upland Papua New Guinea, *Amir* et al. (1990) in Peninsular Malaysia and *Poels* (1987) in Suriname. The information provided by these studies is somewhat more difficult to interpret, however, in view of the non-stratified sampling procedures followed. In the Papuan study, the rather fertile volcanic soil showed a return to near former levels of organic matter after 18 months but concentrations of calcium, potassium, and, to a lesser extent, magnesium were thought to take much longer to recover to pre-logging levels (*Enright*, 1978). No details were given as to where the samples were taken with respect to location of skidder tracks and mounds, etc. On the basis of a random sampling design *Amir* et al. (1990) found rather serious losses of nutrients down to a depth of 30 cm immediately after logging, particularly of calcium, magnesium and nitrogen in the case of fertile (but rather sandy) soils, and of phosphorus in all three soil associations studied. Potassium on the other hand increased, which was ascribed to decompostion of slash and 'parent material' even though the rise in concentration was observed regardless of geological substrate. Recovery after one year was good for nitrogen, reasonable for magnesium, and poor for calcium, whereas potassium remained high. No data were given for phosphorus. The data on changes in soil nutrient concentrations in a selectively logged forest subjected to 'refinement' (poison girdling) in Suriname by *Poels* (1987) were re-calculated by *Bruijnzeel* (1992a) who derived substantial decreases in amounts of calcium and potassium stored in the upper 120 cm of the soil profile. Most of the decreases could be ascribed to vigorous uptake by regenerating vegetation rather than to leaching (*Poels*, 1987; see also below).

Thus far, three studies have attempted to quantify nutrient losses via deep leaching after selectively logging lowland tropical rain forests, viz. *Poels* (1987) in Suriname; *Zulkifli* (1989) in Peninsular Malaysia; and *Brouwer* (1996) in Guyana. The results of the first two studies were discussed in detail by *Bruijnzeel* (1992a). On the bais of the overall nutrient budgets presented by the latter, tentative conclusions were drawn with respect to the time span required for atmospheric contributions of nutrients to compensate the losses associated with timber removal and enhanced leaching (Table 1).

In view of various uncertainties associated with the two studies cited above, the recent work by *Brouwer* (1996) in an area underlain by infertile sandy soils in Guyana, constitutes a welcome addition to the literature. Two gaps of 730 and 3440 m^2, respectively, were created using heavy machinery after which soil water content and composition within and below the main root zone were monitored for almost three years using tensiometry and vacuum tube lysimetry. In addition, soil samples were taken at several points in time according to a stratified sampling schedule. Finally, the chemical compositions of the shallow groundwater and streamflow in a nearby small (6.2 ha) catchment area before and after low-intensity logging (21 m^3 ha^{-1}; *Jetten*, 1994) were monitored as well.

Differences in groundwater levels and stream discharge before and after logging were within the accuracy of the respective measurements and as such the effect of logging, if any, could not be demonstrated at the catchment scale (*Jetten*, 1994). Similar results were previously obtained for lightly logged forests in Queensland (*Gilmour*, 1977) and India (*Subba Rao* et al., 1985). *Bruijnzeel* (1990) concluded from a survey of the literature that at least 20% of the vegetation would need to be removed before any effects on water yield would become noticeable. Changes in the chemistry of both groundwater and stream baseflow in the Guyanese study were negligible as well, with the exception of sodium concentrations which became more variable after logging. *Brouwer* (1996) ascribed this to a combination of the low intensity of the logging (which left a large part of the catchment undisturbed) and a nutrient 'filtering' effect of the vegetation in the riparian zone (cf. *Hsia* and *Horng*, 1990). Apparently, sodium was not in short supply and thus 'permitted' to bypass the riparian zone without being taken up. Some support for this contention may be derived from the results obtained by the Malaysian study referred to already. Here, solute losses in streamflow following a 'commercial' logging operation (removing 40% of the stocking) were two to three times those associated with the 'supervised' harvesting of 33% of the stocking (*Zulkifli*, 1989). The observed difference is hardly explainable in terms of the difference in volume of harvested timber but it remains possible that the retaining of a 40 m wide buffer strip in the 'supervised' case masked the true extent of leaching on the sideslopes (*Bruijnzeel*, 1992a).

As in the Costa Rican example referred to earlier (*Parker*, 1985), the soils in the gaps of the Guyanese study were consistently wetter than those under the adjacent undisturbed forest (leading to approximately 555 mm yr^{-1} of extra percolation during the first 1.8 years; *Brouwer*, 1996). However, rises in soil water nutrient concentrations at 120 cm depth were both more pronounced (up to 20 times higher than observed under undisturbed forest in the case of nitrate and 4–10 times for calcium, magnesium and potassium) and commenced earlier (within a month after disturbance) than in the Costa Rican experiment. Such differences probably reflect the lower water and nutrient retention capacity of the sandy soils in Guyana. Although concentrations started to decrease again 12–15 months after gap creation they generally remained stable at levels that exceeded those observed in the undisturbed plots until at least 34 months after disturbance (*Brouwer*, this volume; cf. *Uhl* et al., 1982). Total amounts of nutrients leached from the two gaps during the entire period were substantial, with losses of nitrogen, calcium, magnesium and potassium from the larger gap being *ca.* 17, 5, 10 and 4 times those observed under undisturbed forest (*Brouwer*, this volume). The corresponding leaching losses from the smaller gap were roughly 50% of those associated with the bigger gap, except for calcium losses which were similar in both cases (*Brouwer*, 1996; Table 1). These results suggest that the contention of *Parker* (1985) that the amounts of nutrients leached from his two large gaps were very similar, may well be in need of modification (cf. *Brouwer*, this volume). Further work is necessary.

In view of the virtual absence of weatherable minerals in the substratum of the gaps in the Guyanese study, replenishment of the nutrients lost via increased leaching and timber extraction will have to come again primarily from the atmosphere (except for nitrogen). Approximate estimates for the corresponding periods required to compensate the respective losses have also been included in Table 1. On the basis of the results presented in Table 1 it remains to be seen whether the very high intensities of timber extraction that are common in parts of Borneo (up to 120 m^3 ha^{-1}; *Whitmore*, 1990), may not lead to an unacceptable degree of ecosystem nutrient depletion. Serious decreases in nutrient reserves would certainly occur after a few rotations in areas where contributions of rock weathering are negligible. One could argue that soils in steep terrain tend to be shallower and the weathering front closer to the surface and therefore whithin easier reach of the roots (*Baillie* and *Mamit*, 1983; *Burnham*, 1989). Under such conditions, nutrients released by weathering could theoretically constitute an important component of the nutrient budget. Naturally, actual amounts so contributed will depend on rainfall totals and geological substrate (*Bruijnzeel*, 1991). The observations of *Grip* et al. (1994) at Sipitang, Sabah, indicate that amounts of calcium, magnesium, and potassium released by the weathering of sandstones amount to only a few kg ha^{-1} yr^{-1} but that the corresponding amounts produced by shales (10–20 kg ha^{-1} yr^{-1}) would be sufficient.

The information compiled in Table 1 suggests that the prime loss of nutrients upon logging tropical forest occurs in the form of harvested timber rather than through enhanced leaching. Nevertheless, there is a need to expand the data set on the various gains and losses of nutrients associated with logging operations for different harvesting regimes and geological substrates (cf. *Bruijnzeel*, 1992a; *Nykvist*, 1992). Similarly, published information on the changes in streamflow totals after logging is scarce and contradictory (see *Bruijnzeel* (1996) for a detailed discussion).

In contrast to the gaps created by (selective) logging, which tend to be of limited areal extent, some truly massive gap creation c.q. forest disturbance in the form of canopy defoliation, the snapping or uprooting of trees as well as landsliding, may occur during the passing of tropical hurricanes (*Scatena* and *Larsen*, 1991; *Walker*, 1991; *Scatena* et al., 1993; *Waterloo*, 1994). The sudden addition of large masses of fresh organic material to the forest floor, coupled with dramatic increases in light levels, soil temperatures and soil water content, bring about an array of changes in the processes governing release, cycling and loss of nutrients, particularly nitrogen (*Lodge* et al., 1991; *Blood* et al., 1991; *Steudler* et al., 1991). For example, ammonium availability, net nitrogen mineralization and nitrification rates were all observed to be elevated four months after the passage of Hurricane Hugo in Puerto Rico (*Steudler* et al., 1991). Similarly, concentrations of nitrate and base cations in streamwater also increased, although overall solute losses remained low due to the rather small amounts of streamflow during this period (F. N. Scatena, personal communication). *Waterloo* (1994) also reported significant changes in streamflow chemistry during the first month after extensive damage occurred to a pine forest estate in Viti Levu, Fiji during the passage of a cyclone (Sina). Interestingly, ammonium and nitrate concentrations were not affected here, while concentrations of base cations increased and levels of sulphate were reduced. *Waterloo* (1994) explained the latter finding in terms of the chemical composition of the rainfall and sea spray associated with the cyclone which happened to be low in sulphate. No reduction in sulphate concentration was observed in the flow from an adjacent grassland catchment which may reflect the lower trapping efficiency of the grass for cyclone sea spray (*Waterloo*, 1994). The lack of response of the nitrogen constituents may be related to the stimulation of nitrogen immobilization by the microbial community through the large addition of fresh organic matter via needle fall during the cyclone (cf. *Vitousek* and *Matson*, 1985).

Despite the potentially more widespread areal impact of hurricanes, *Steudler* et al. (1991) considered the effects of *forest clearcutting* on the soil nitrogen cycle and trace gas fluxes to be more intense and of longer duration. In the hurricane case, the root system of much of the (natural) vegetation generally remains in place and many trees and seedlings respond rapidly (*Brokaw* and *Walker*, 1991). As a result, the nitrogen cycle can be expected to recover quickly, whereas the vegetation is slower to re-establish itself after clear cutting, particularly so after the use of heavy machinery (*Uhl* et al., 1982, 1988a; see also the next section). *Malmer* and *Grip* (1994) carried out an interesting experiment in this respect in Sabah, Malaysia. They monitored the flow and solute losses from an entire catchment area (W_4, 3.4 ha) before, during and after it was subjected to clearcutting and timber extraction. Some 146 m^3 of timber was removed from the area using manual labour to minimize surface disturbance (*Malmer* and *Grip*, 1990; *Sim* and *Nykvist*, 1991). The remaining slash was left on the slopes which were then planted to *Accacia mangium*. During the first 1.5 and 3.7 years after planting the *Acacia* trees acquired above-ground biomass figures of 10.5 and 44.6 t ha^{-1}, respectively (*Malmer*, 1993). The nutrient losses associated with timber extraction and leaching have been included in Table 1 for comparison. With the exception of potassium, which was leached severely, overall losses for the other macro-nutrients were modest and could theoretically be recovered within 35–55 years by atmospheric inputs alone and within 5–20 years once contributions of rock weathering are taken into account as well (Table 1).

Forest fires and shifting cultivation

The occurrence of fire may increase overall site nutrient losses in various ways. Firstly, there are the losses associated with volatilization and ash particle transport during the burn itself (*Raison* et al., 1985; *Ewel* et al., 1981;

Mackensen et al., 1996). Secondly, nutrients contained in the ash residue will be more prone to leaching and erosion (by both water and wind) than those in gradually decomposing slash (*Toky* and *Ramakrishnan*, 1981; *Uhl* and *Jordan*, 1984; *Waterloo*, 1994). Finally, surface erosion may increase dramatically after burning (*Wiersum*, 1984). Changes in soil physical characteristics due to fire range from negligible to dramatic (*Van Lear* et al., 1985; *Leitch* et al., 1983). Much depends on the intensity of the fire which is governed mainly by amounts and moisture content of the material available as fuel (undergrowth, logging debris and litter) and ambient weather conditions (*Scott*, 1993; *Mackensen* et al., 1996). Different soils may exhibit a different response to excess heat, with some soil types even becoming temporarily water repellent. Naturally, water repellence of soils impedes infiltration, and this may lead to the generation of overland flow (*Burch* et al., 1989; *Scott*, 1993). When the overland flow occurs on sites where the protective litter layer was just destroyed by the fire, some truly dramatic surface and gully erosion may follow (*Brown*, 1972; *Leitch* et al., 1983). However, such effects are hard to distinguish from those exerted by soil compaction.

Nutrient losses to the atmosphere during the burning of slash in tropical regions have been quantified by only a few studies (Table 2), some of which (e.g. *Toky* and *Ramakrishnan*, 1981) seem to have produced anomalously high values. Contrary to earlier assumptions (e.g. *Ewel* et al., 1981), losses due to volatilization do not merely concern those elements having a gaseous phase (nitrogen, sulphur) but also the base cations calcium, magnesium, potassium as well as phosphorus (Table 2). Whilst some of the higher figures quoted for the respective cations in Table 2 may be due to wind effects (*Waterloo*, 1994), the ones pertaining to the Amazonian study were obtained under conditions of still air and must therefore represent upward molecular transport related to thermally induced kinetic effects (*Mackensen* et al., 1996). In contrast to losses of nitrogen and sulphur, those for calcium etc. appear influenced by the amount of fuel and the intensity of the burn (*Mackensen* et al., 1996; Table 2).

There have been numerous studies of the changes in soil chemical contents at the respective stages of the *shifting cultivation* cycle (see reviews by *Sanchez*, 1976; *Bruijnzeel*, 1990; *Ramakrishnan*, 1992). Such changes are the net result of an array of nutrient inputs and outputs, of which volatilization and erosion, uptake by regenerating vegetation, and leaching losses are generally considered to be the most important (*Uhl* and *Jordan*, 1984, *Wiesum*, 1984; *Ramakrishnan*, 1992). The lack of reliable estimates of the amounts lost via enhanced leaching after tropical forest cutting and burning has been commented upon already. As such, the results of an ongoing study of the water and nutrient dynamics of freshly cleared and burned as well as regenerating sites in eastern Amazonia using a combination of micro-meteorological, plant physiological and soil hydrological techniques (*Hölscher*, 1995; *Mackensen* et al., 1996; *Roberts* et al., 1996; *Klinge*, 1997) are a welcome addition to the literature. More work is needed, however, covering different combinations of rainfall regime and soil type.

There are virtually no data on the hydrological impacts (including enhanced solute losses) of *natural forest fires* at the catchment scale in the humid tropics for the simple reason that river gauging stations (if present at all) are usually destroyed by such fires, and their re-establishment tends to take longer than that of a vegetative cover (*Brown*, 1972). Arguably, the best way of collecting such information is through experimental studies (*O'Loughlin* et al., 1982). A particularly interesting case has been reported by *Malmer* (1993) for experimentally burned secondary scrub (biomass < 5 t ha^{-1}) in Sabah that had been hit by a forest fire 5–6 years previously. After the experimental burn, nutrient concentrations in streamwater increased more vigorously in catchments that had experienced the great fires of 1982–83 than for a mechanically logged catchment where far greater amounts of biomass (logging debris) were burned (see Table 3 below). The effect was much more short-lived, however, and within 6 months concentrations in the basins that had been burned twice started to fall below predicted levels. *Malmer* (1993) interpreted

Table 2: Relative nutrient losses (per cent) due to volatilization during the burning of residual biomass in selected tropical forest areas as a function of fuel mass (t ha^{-1}) and mass reduction (per cent).

Location	Fuel mass (t ha^{-1})	Mass reduction (per cent)	N	P	K	Ca	Mg	S
			(per cent)					
Turrialba, Costa Rica[1]	38	83	23		negligible?			44
Belém, Brazil[2] site 2	33	90	95	26	16	9	17	67
Belém, Brazil[2] site 3	95	96	98	33	31	25	43	68
Viti Levu, Fiji[3]	40	86	84	52	79	78	60	–

[1]Ewel et al. (1981);
[2]Mackensen et al. (1996);
[3]Waterloo (1994).

Table 3: (a) Nutrient losses (kg ha^{-1}) in harvested timber and (b) deep leaching after clearcutting and/or burning tropical forest; (c) approximate time span (yr) required to compensate the losses via nutrient inputs from atmospheric sources only, and (d) *idem* plus rock weathering; values of estimated time spans above 10 yr rounded off to the nearest 5.

Location	Treatment	Harvested timber (t ha^{-1})	Burned slash (t ha^{-1})		N	P	K	Ca (kg ha^{-1})	Mg
Sipitang, Malaysia[1–3@]									
Basin W_{1+2}*	Burning only	–	5	(a)	–	–	–	–	–
				(b)	17	2	84	28	–5
				(c)		6	15	10	–
				(d)		–	9	2	–
Basin W_5#	Mechanized logging + burning	75	170	(a)	80	2	57	112	32
				(b)	40	1	189	27	16
				(c)		10	50	50	50
				(d)		–	35	15	6
Basin W_4°	Manual logging + no burn	90	185	(a)	100	2	71	139	39
				(b)	27	1	106	25	8
				(c)		10	35	55	45
				(d)		–	20	10	4
Viti Levu, Fiji[4+]	Mechanized logging + burning	55	40	(a)	45	5	29	23	11
				(b)	6	< 1	3	2	4
				(c)		20	10	15	6
				(d)		9	6	1	< 1

[1]Sim & Nykvist (1991);
[2]Grip et al. (1994);
[3]Malmer & Grip (1994);
[4]Waterloo (1994);
@high rainfall environment (3350 mm yr^{-1}); paired catchment technique; leaching losses computed for a period of 34 months; weathering contributions estimated as indicated in Table 1;
*6.45 ha catchment; 38 per cent podzols, 62 per cent ultisols; vegetation burned five years prior to the experiment and since then consisting of scrub, grasses and ferns;
#9,67 ha catchment; 62 per cent spodosols, 38 per cent ultisols; selectively logged about 10 years before experimental cutting and burning;
°3.4 ha catchment; 34 per cent spodosols, 66 per cent ultisols; selectively logged about 10 years before experimental logging;
+single catchment experiment; low rainfall environment (1800 mm yr^{-1}; leaching losses for the first 16 months only; basin size 62,9 ha; moderately fertile ultisols; basin 18 per cent covered by native riparian vegetation and remainder by 15-year-old *Pinus caribaea*; 42 per cent of plantation damaged by hurricane in November 1990; intermittent tractor logging between December 1990 and July 1991; burned in August 1991; weathering contributions calculated from catchment nutrient input-output budget, taking into account amounts immobilized in biomass and assuming no net changes in soil nutrient stores during a rotation.

this as an exhaustion effect. However, the phenomenon was observed most strongly for silica and it would seem that the reported changes in streamwater solute concentrations reflected a shift in water pathways (lowered baseflows and increased peakflows) following burning rather than just nutrient uptake processes alone. Such findings illustrate the need for integrated process studies to illuminate the traditional paired catchment approach which, although statistically sound, remains essentially a black box approach (*Grip*, 1994; *Bruijnzeel*, 1996). More work is needed with respect to the fate of nutrients released by the burning of residual biomass (see also below).

3.2 Response to disturbances of high intensity (forest conversion)

It is well-established that manual methods of forest clearing are superior to most wheeled or tracked equipment in terms of damage to the forest floor (*Lal*, 1987; *Fritsch*, 1992; *Bruijnzeel*, 1996). Indeed, there are strong arguments for the use of alternative mechanized timber extraction methods that tread lightly on the soil surface, such as skyline yarding (*Ludwig*, 1992), elephants where the size of the logs allow this (*Whitmore*, 1990), or even helicopters in the case of well-stocked forests in steep terrain (*Blakeney*, 1992). As will be shown below, there is also increasing evidence that leaving slash on site to rot rather than 'windrowing' and burning it has various beneficial effects on the hydrological and chemical characteristics of the soils. However, when converting rain forest land to other land use types, the use of heavy mechanized equipment is often unavoidable. *Martin* (1970) and *Couper* et al. (1981) have offered various suggestions as to how damage to the soil can be minimized.

Naturally, nutrient losses associated with large-scale mechanized forest clearfelling and burning large amounts of residual biomass are potentially much greater than those for the less intensive types of forest disturbance discussed in the previous sections, particularly under conditions of high rainfall. A review of changes in water yield following tropical forest clearance by *Bruijnzeel* (1990) revealed that reported first-year increases in streamflow ranged between 125 and 820 mm. There can be little doubt that such increases reflect the different evaporative characteristics of mature tropical forest and young secondary or planted vegetation, and, therefore, differences in soil water status (cf. *Parker*, 1985; *Brouwer*, this volume). Interestingly, the large variation in initial streamflow response to clearing can be explained only partially by differences in rainfall between locations (Figure 3). Other factors include differences in elevation (and thus evaporation), catchment steepness and soil depth (govering the residence time of the water in the catchment, baseflow recession and stormflow patterns; *Ward*, 1984), and above all the degree of disturbance of the soil by machinery or fire (influencing runoff patterns; *Malmer* 1992; *Swindel* et al., 1982, 1983), and the fertility status of the soil (affecting, like the intensity of soil disturbance, post-clearing plant productivity; *Uhl* et al., 1982, 1988a; *Brown* and *Lugo*, 1990). The relative importance of the respective factors will vary between locations and the resulting high variability of the data (Figure 3) does not allow any detailed quantitative predictions. Again, additional process studies (particularly on runoff generation mechanisms) are required if the results of black box catchment experiments (Figure 3) are to be more fully understood (*Bonell* and *Balek*, 1993; *Bruijnzeel*, 1996).

Most of the factors listed above also affect the amounts of nutrients lost via erosion and deep leaching, illustrating the close links between water and nutrient pathways (*Likens* et al., 1977). As noticed earlier for studies of small-scale forest cutting and burning (*Toky* and *Ramakrishnan*, 1981; *Uhl* et al., 1982; *Uhl* and *Jordan*, 1984), most early investigations of the effects of tropical forest conversion on soil and ecosystem nutrient reserves (*Hase* and *Fölster*, 1983; *Russell*, 1983; *Buschbacher*, 1984) used a 'false time series' approach. Leaching losses were generally estimated on the basis of rather flimsy hydrological budgets or simply as average values taken from the literature while nutrient losses via surface erosion were usually not taken into account either (*Bruijnzeel*, 1990). Despite the shortcomings of such studies, their overall conclusions as to which macro-nutrients were likely to become limiting first, are probably justified, although some authors (e.g. *Hase* and *Fölster*, 1983) were quick to point out that their nutrient budgets should be regarded as preliminary and semi-quantitative.

Table 3 summarizes the losses of nutrients in harvested timber and enhanced leaching as determined by two recent studies of mechanized forest clearing and burning of slash under contrasting environmental conditions. The Malaysian site experiences high annual rainfall totals (> 3300 mm; *Malmer*, 1992) while the Fijian site is characterized by relatively low rainfall (c. 1800 mm yr^{-1}) and a long dry season (*Waterloo*, 1994). Also illustrated in Table 3 are the contrasting effects of (i) burning different amounts of slash (catchments W_{1+2} vs. W_5) and (ii) mechanized clearing plus burning (catchment W_5) vs. manual clearing and leaving slash to rot (catchment W_4).

Despite the severity of the treatment, amounts of nutrients lost from the Fijian catchment during the first 16 months after (pine) forest clearcutting and burning some 40 t ha^{-1} of slash were small. Various explanations may be offered for this finding, including below-average rainfall totals during the first rainy season after the burn; the filtering effect of the riparian vegetation which remained undisturbed (cf. *Zulkifli*, 1989; *Brouwer*, 1996); the vigorous regrowth of the understorey vegetation (cf. *Uhl* and *Jordan*, 1984); and the high nutrient retention capacity of the clayey subsoils in the area (*Waterloo*, 1994).

As could be expected on the basis of the higher rainfall and/or larger amounts of burned residual biomass,

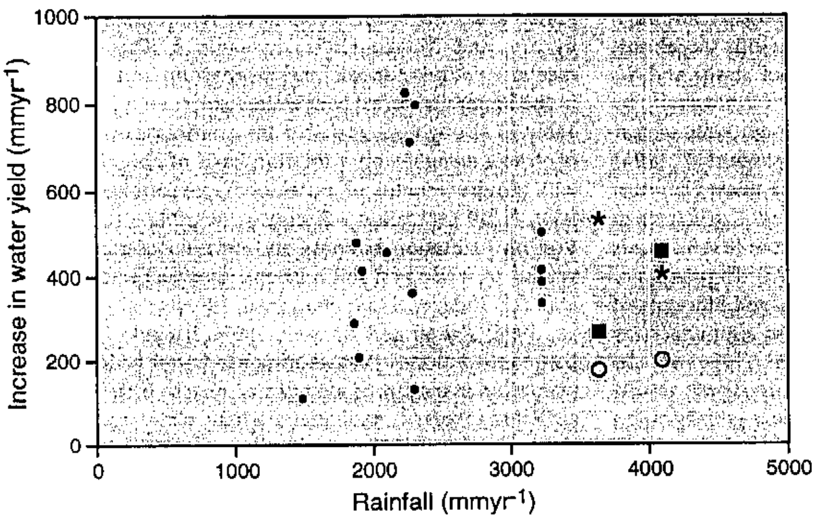

Figure 3: Increases in water yield (mm yr^{-1}) during the first three years after clearing tropical forest vs. corresponding amounts of annual rainfall (after Bruijnzeel, 1997). ○ catchment W_4 (manual clearing), ■ catchment W_5 (mechanized clearing plus burning of slash), and * catchment W_{1+2} (burning only) of Malmer (1992).

cationic losses in solution were generally much larger in the Malaysian experiment (even when allowing for the longer observation perid, 34 vs. 16 months; Table 3). This was the case regardless of the treatment, with the exception of magnesium losses after logging only, which were similar to those observed in the Fijian study (Table 3). As indicated earlier (section 3.1), the negative result obtained for magnesium after burning only (catchment W_{1+2}) probably reflects a shift in runoff patterns. Unfortunately, it is not known to what extent this phenomenon affected leached amounts of calcium and potassium as well (*Malmer* and *Grip*, 1994). Leaching of calcium was similar for all three treatments (Table 3), despite large differences between catchments in water yield after conversion (more than twice as high for W_{1+2} and W_4 compared to W_5; *Malmer*, 1992) and a different soils association in catchment W_5. Solute losses of potassium were about twice as high after mechanized harvesting and burning of slash (W_5) in comparison with either burning (W_{1+2}) or manual logging only (W_4). Similarly, magnesium losses in streamflow after the most severe treatment (W_5) were also double those associated with logging only (W_4) (Table 3).

Strictly speaking it is difficult to draw any firm conclusions on the basis of these results with respect to solute losses as a function of treatment, other than that additional process work is still required. For example, a direct comparison of the leaching losses obtained for catchments W_5 (tractor logging + burning) and W_4 (manual logging only) is hampered because of the different proportions of sandy and clayey soils found in the respective basins. The overall water and nutrient retention capacities of the predominantly sandy soils in W_5 could be expected to be lower than those of the predominantly clayey soils in W_4 (*Grip* et al., 1994; cf. footnotes to Table 3). Also, the similarity in losses of calcium, but not in magnesium or potassium, for the two catchments is difficult to explain without additional information on soil nutrient reserves and nutrient uptake by the vegetation. Similarly, although the areal distribution of sandy and clayey soils was almost identical for catchments W_{1+2} (burning of slash) and W_4 (rotting of slash), the effect of the burn cannot really be predicted because of the vastly differing amounts of residual biomass involved (Table 3; cf. *Mackensen* et al., 1996). A further complicating factor concerns the related contrast in growth exhibited by the understorey and *Acacia* trees in the two areas (22.2 vs. 50.0 t ha^{-1} during the first 3.5 years in W_{1+2} and W_4, respectively; *Malmer*, 1993). The results of the inventories of soil chemical reserves and amounts of nutrients incorporated in forest biomass announced by *Sim* and *Nykvist* (1991) and *Nykvist* et al. (1994) will be needed to obtain a more complete picture. These results illustrate once more the importance of close collaboration between pedologists and hydrologists when it comes to experimental design and site location (see also *Fritsch* et al., 1987; *Fritsch*, 1993).

Nutrient losses associated with the harvesting of stems were generally greater than those in streamflow in both studies, with the notable exception of potassium which was leached in large quantities from the Malaysian catchments (Table 3). All in all, cationic losses in the latter environment were such that at least 35 to 55 years of nutrient additions via bulk precipitation would be required to compensate them (Table 3). The corresponding recovery periods for the respective elements in the Fijian case, where both leaching losses and volume of harvested timber were lower, ranged from about five years (magnesium) to 15 years (calcium).

The information presented in Table 3 may be used to help assess the eventual effects of plantation forestry on soil nutrient reserves. Overall nutrient inputs from atmospheric sources throughout a typical pine plantation rotation period of 15–20 years in the Fijian example were such that they would cover the losses associated with stemwood (plus bark) removal and temporarily increased leaching (*Waterloo*, 1994). Because amounts of nutrients (particularly calcium and magnesium) released by rock weathering in this area were substantial and the weathering front usually within reach of the pine root network (*Waterloo*, 1994) one may conclude that the cultivation of *Pinus caribaea* in south-western Viti Levu will not lead to unacceptable site degradation. However, *Waterloo* (1994) added the precaution that both whole-tree harvesting and surface erosion should be avoided.

It is well-established that nutrient concentrations in the stemwood and bark of non-coniferous species (particularly teak and *Gmelina*) are often much higher than those contained in most pine species (*Halenda*, 1993; *Fölster* and *Khanna*, 1997). It follows that amounts of nutrients lost from the site in harvested hardwood timber are potentially much greater than for pines or, indeed, natural forest. For example, amounts of nutrients accumulated in the stems of a 6.6-year-old stand of *Gmelina arborea* on a very acid ultisol in Sarawak (*Halenda*, 1993) exceeded those contained in the harvested stems of the rain forest clearfelled by *Sim* and *Nykvist* (1991) in Table 3. The estimated recovery periods for calcium, magnesium and potassium in the Malaysian forest conversion experiment (c. 50 years, Table 3) strongly suggest that the continuous production of *Acacia mangium* on a rotation basis of about ten years will require additional nutritional measures sooner or later. A full evaluation will only be possible after information on plantation biomass and nutrient content at harvesting as well as on soil nutrient reserves becomes available (cf. *Ruhiyat*, 1989; *Bruijnzeel*, 1992b; *Spangenberg* et al., 1996; *Fölster* and *Khanna*, 1997). It could be argued that contributions by weathering had better be ignored by plantation managers as long as we know so little about the amounts involved, particularly where trees are grown in short rotations (*Fölster* and *Khanna*, 1997). The latter authors did recognize a role for weathering, however, in the case of plantations with longer rotations (> 20 years) and certain soil groups that still contained weatherable minerals. Nevertheless, the provisional estimates of the amounts of nutrients released by rock weathering in the Malaysian example (notably of the shales in catchment W_3) suggest that recovery periods could be substantially reduced by the inclusion of nutrient inputs from weathering, provided the trees would have access to them (Table 3). The quantification of the amounts of nutrients released by mineral weathering is notoriously difficult (*Clayton*, 1979; *Verstraten*, 1980) and its estimation has only rarely been attempted within the framework of tropical forestry (*Bruijnzeel*, 1983; *Hase* and *Fölster*, 1983; *Tandy*, 1987; *Grimaldi*, 1988). Yet this aspect may be expected to become increasingly important in view of the rapidly expanding area of fast-growing forest plantations in the humid tropics and the high costs of fertilizer (*Evans*, 1992).

4 Research perspectives

Several important gaps in our knowledge have become apparent in the previous discussion. Arguably, the most striking of these is the *almost complete lack of reliable information on the hydrological behaviour (rainfall interception, transpiration, percolation) of young secondary vegetation and tree plantations* (*Hölscher*, 1995; *Bruijnzeel*, 1996, 1997). Needless to say, such information is a *sine qua non* for the estimation of amounts of nutrients lost via leaching. As shown in Tables 1 and 3, leaching losses upon forest disturbance and conversion can be substantial, particularly under conditions of high rainfall and soils with low nutrient retention capacity. The methodology for the estimation and modelling of rainfall interception and water use is well-established (*Lloyd* and *Marques*, 1988; *Lloyd* et al., 1988; *Roberts* et al., 1993, 1996), although the scattered nature of many secondary stands in the tropics (*Uhl* et al., 1998a; *Denich*, 1989) may present methodological problems (*Hölscher*, 1995). A useful role may be played in this respect by such approaches as the isotope tracer injection technique (*Calder*, 1991) and the heat pulse velocity (or heat balance) method (*Hatton* and *Vertessy*, 1990) which may (also) be used to study water uptake rates at different topographic positions (e.g. dry ridge tops vs. wet valley bottoms) or from soils of different degrees of compaction.

Although amounts of slowly percolating soil water (unsaturated matrix flow) can be quantifed adequately using the various techniques listed above, additional observations will be required of the amounts and chemical composition of the rapid flow through macropores and pipes during periods of intense rainfall if total amounts of leached nutrients are to be determined accurately. Indeed, the *generation of runoff during storms (both at and below the surface) constitutes one of the more poorly documented aspects of tropical forest hydrology* (*Bonell* and *Balek*, 1993; *Elsenbeer* and *Lack*, 1996; *Klinge*, 1997). Naturally, the study of such 'bypass flow' or 'quickflow' (depending on one's standpoint: the soil profile or the stream/catchment) assumes particular importance in relation to the proper understanding of the effects of forest disturbance or conversion. *Bishop* (1991) and *Bonell* (1993) have provided recent in-depth discussions of the direction that future runoff generation investigations could take. Both stressed the need for an integrated approach that would combine classical hydrological (volumetric) techniques and detailed event sampling (for solutes and stable isotopes in soil- and streamwater) with physically-based distributed hydrological models. Because of the great depth to bedrock in many tropical catchments (*Burnham*, 1989), however, the application of such models may not be as straightforward as it looks (*Quinn* et al., 1991). *McDonnell* et al. (1996), working on deeply weathered soils in the south-eastern U.S., recently demonstrated the close correspondence between subsurface flow paths and bedrock topography. Similar studies under humid tropical conditions could be revealing.

There is also a *need for more studies of nutrient losses via eroded topsoil material after forest clearance and burning as a function of rainfall regime, soil type and degree of disturbance* (*Hudson* et al., 1983; *Bruijnzeel* and *Wiersum*, 1985; *Malmer*, 1996). Ideally, such work should be conducted at various levels of scale (hillslope plots vs. small catchments) so as to enable the assessment of on-site (erosion) and off-site (basin sediment yield) effects. Such experiments could be usefully *combined with research into the filtering effects of streamside buffer strips* whose role with respect to nutrient interception is still poorly defined (*Zulkifli*, 1989; *Hsia* and *Horng*, 1990; *Brouwer*, 1996). These aspects assume particular importance in the context of fast-growing forest plantations with short harvesting cycles (< 5–8 years) and, consequently, frequent surface disturbance (*Wiersum*, 1984; *Fölster* and *Khanna*, 1997).

Finally, the *potential relevance of nutrient inputs to tropical forest ecosystems by mineral weathering* has already been commented upon in previous sections. Research to this end is probably conducted best via the small catchment nutrient budget approach (*Clayton*, 1979), supplemented by water-rock interaction studies (*Bruijnzeel*, 1983; *Tandy*, 1987; *Grimaldi*, 1988). Such work could well be combined with the study of the subsurface topography of the fresh bedrock in relation to hillslope hydrological pathways and nutrient export routes (*McDonnell* et al., 1996).

Evidently, there are distinct opportunities for closer collaboration between tropical pedologists, hydrologists and plant ecologists. Thus far, no single study has addressed the soils and water aspects of a tropical forest ecosystem in a fully integrated manner, either in undisturbed forest or after disturbance or conversion (*Bruijnzeel*, 1983; *Fritsch*, 1992; *Malmer*, 1993). In view of the very considerable investments in manpower and time required by such studies (*Likens* et al. 1977; *Swank* and *Crossley*, 1988) it would seem advisable to focus future integrated research efforts on a limited number of sites for which detailed soils and vegetation inventories are already available and where the basic fluxes of water and nutrients have been measured for a sufficiently long time (cf. *Bruijnzeel* and *Abdul Rahim*, 1992). Similarly, the renewed assessment of amounts of nutrients currently stored in the soil and vegetation of sites that were the subject of investigation in the past could prove more effective than the establishment of entirely new locations. Examples include the teak forests in southwestern Venezuela studied by *Hase* and *Fölster* (1983) or the pine plantations in Fiji sampled by *Waterloo* (1994). Since there are a number of existing research sites scattered all over the humid tropics (*Bruijnzeel*, 1990), there may be scope for a screening exercise on the basis of which decisions could be taken as to where additional sites would be established best (*Bruijnzeel* 1993, 1996). For instance, there are virtually no studies evaluating the hydrological and fertility aspects of reforesting degraded areas at the catchment scale (cf. *Waterloo*, 1994), or the impacts of clearing montane cloud forests (*Bruijnzeel* and *Proctor*, 1995). Arguably, the coordination of such a (possibly rather informal) pantropical 'network' of research sites might be handled best by the Humid Tropics Programme within UNESCO's International Hydrological Programme.

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