SWEDISH UNIVERSITY OF
AGRICULTURAL SCIENCES
DEPARTMENT OF FOREST ECOLOGY

Dynamics of hydrology and nutrient losses as response to establishment of forest plantation.

A case study on tropical rainforest land in Sabah, Malaysia.

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Cover photos by the author

Front left: Surface runoff during rain in December 1991 on a four years old tractor track in catchment W5 with clear-felling, crawler tractor extraction and burning before planting. Note the stunned growth of the *Acacia mangium* planted on the track and the slow recovery of the track top soil. Hardly no plants were still rooted in the very track but only expanding from the sides.

Front right: Extensive gully erosion on tractor track 2 (Table III, paper VI) on the 14th of December 1988 in catchment W5 during clear-felling and tractor extraction. Compare streamwater baseflow concentration in Figure 6, page 28.

Back top: Catchment W5 after clear-felling, crawler tractor extraction and burning, before planting in March 1988. Part of catchment W4 with manual extraction and no burning is visible in the far left background. Note the large amount of biomass left in slash.

Back bottom: 7 months old *Acacia mangium* in cleared planting rows in the unburned slash in catchment W4. The blue canvas indicate the collecting gutters for runoff plots "M" in W4 (cf. paper VI).

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ABSTRACT

The objective of this thesis was 1) to quantify effects on runoff, erosion and dissolved streamflow nutrient losses when clear-felling tropical rainforest with or without soil disturbance and subsequent burning of residues before the establishment of forest plantations of *Acacia mangium*, 2) to increase the transferability of these and other results by explaining differences in streamwater nutrient losses after different treatments from understanding of the interaction of erosion-, leaching- and transport processes. The work of this thesis is included in a larger research project carried out as a paired catchment study within the research area in Mendolong, Sabah, Malaysia.

Dissolved nutrient losses were enhanced by; A) the amount of nutrient rich parts in the residues under mineralization and leaching during a short time. B) lack of vegetation to use released nutrients and C) rate of runoff increase, especially in surficial flow paths. The most extreme total net losses were induced by normal practice of tractor extraction and subsequent burning. Both soil disturbance and burning optimized A, B and C. For this treatment, resulting losses were substantial, like N-tot 39.9, P-tot 1.3 and K 189 kg ha^{-1} during the first 2.5 years, in the case of K larger than the removal in the harvest.

The largest dissolved loss rate and absolute loss during and after burning (like 131 kg ha^{-1} of K in 5 months) were recorded from burning of secondary vegetation after forest fire. This was because of a hard burn combined with no soil disturbance, leaving undisturbed surficial flow paths optimizing transport to the stream.

Erosion on undisturbed clear-felled soils were in the size of undisturbed forests due to maintained surface protection and aggregated top soil with high infiltrability and top soil hydraulic conductivity. However, the quality aspect of lost sediments were pointed out as a small increase in slope surface erosion of ashes after burning might have carried large amounts of nutrients. Soil disturbed by crawler tractors on the other hand experienced severe gully erosion, stream siltation, prolonged runoff increases, by increase of the length of ephermal channels, and finally very slow recovery of favorable top soil properties.

A "minimum disturbance treatment" with manual extraction and no burning reduced effects of increases in runoff, stream siltation and dissolved nutrient net losses to about 50 %. This was due to the reduction by all three points above; A) slower mineralization and leaching from large amounts of residues by avoiding effects of soil disturbance and concentration of nutrients to ash, B) and C) more vegetation left living and faster initial response by secondary growth due to no disturbed soil surfaces and no burning.

key words: tropical rainforest conversion, forest plantation, *Acacia mangium*, logging effects, harvesting technique, clear-felling, water balance, runoff increase, nutrient budget, nutrient losses, leaching, soil physical properties, soil disturbance, slope hydrology, erosion, water quality, Sabah, Malaysia.

In memory of my father

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2. INTRODUCTION AND OBJECTIVES

2.1 Introduction

The resource of the managed natural forest and the use of extractive tree plantations are becoming increasingly important due to the depletion of the forest resource in the tropics (Whitmore, 1990). Consequently forestry was one of the main issues at the UNCED conference in Rio de Janeiro 1992 (eg. French, 1992). Logging and clear-felling of tropical rainforest cause disturbance to soil and water. This disturbance will vary in amount and duration depending on intensity of the operation and the methods used (Hamilton and King, 1983). To secure sustainability of future forest production it will be increasingly important to optimize silvicultural systems, for selective logging and for land clearing for forest plantations, in the respect of minimizing negative impacts on soil and water (eg. Dykstra and Heinrich, 1992; Mok, 1992). Already Nye and Greenland (1964) found large differences between nutrients lost from top soil and much smaller removal in harvest, which they connected to leaching loss, after two years of cropping in Ghana. Processes of erosion and changes in hydrology have been studied for a long time on the plot scale as well as the catchment scale. However, to understand the dynamics of several different combined disturbances (clear-felling, soil disturbance and burning) in different parts of a catchment and the effect on nutrient dynamics and site sustainable production, different approaches and scales have to be combined (Anderson and Spencer, 1991). Few such combined approach studies has been published from tropical forests (Bruijnzeel, 1990).

The present paired catchment study, started in 1985, is monitoring the hydrological, hydrochemical, soil and biomass changes before, during and after different ways of converting selectively logged tropical rainforest and forest struck by forest fire to forest plantation. Quantification of effects on the catchment scale were combined with detailed studies of soil physical and chemical properties as well as on slope hydrological processes and biomass investigations.

This thesis includes a description of the control forest ecosystem and there after a report on consequences of different measures included in the forestry treatments. The emphasis in these two parts are put on water balance, slope hydrological processes and nutrient balance. The control ecosystem description includes brief reviews of these three subjects as well as erosion and slope solute removal with emphasis on the humid tropical forest environment. Se further the paragraph 2.3 on objectives below.

2.2 *Background*

When the first integrated pulp- and paper mill in Malaysia was projected in Sipitang, Sabah, some impact studies were started as a prerequisite. These were impact studies concerning the marine ecology of the Brunei Bay, the siltation of the lower part of the large river Sungai Mengalong and ecological consequences from the forestry operations. The study on environmental consequences of clear-felling of tropical forest land was started 1984 by Ångpanneföreningen - Industrins Processkonsult (ÅF-IPK) as a consultancy mission given by the State Government-owned Sabah Forest Industries Sdn Bhd (SFI) with the Department of Forest Ecology (formerly: Department of Forest Site Research) of the Swedish University of Agricultural Sciences (FE) as under consultant. Since 1987 this project has been run as a joint project between SFI and FE, funded to equal parts by Swedish Agency for Research Cooperation with Developing Countries (SAREC) and SFI. Since 1990 the joint partnership is between FE and Forest Department of Sabah.

The research area was chosen in late 1984. It was chosen to be representative to other forests in the area being converted to plantation forest. Therefore it was set up in forest lightly selectively logged in the early 80's and in intermediate elevation compared to areas planned for conversion to forest plantation. It was also chosen to include areas which were struck by the extensive forest fires in 1982/83 (Woods, 1989), as these areas struck by fire were chosen in the first place to reforestate with the forest plantations.

This thesis includes the effectuation of study and analysis of the paired catchments as to the water balance and physical and chemical water quality between 1985/1986 to 1990. Furthermore also plot and point studies on soil physical properties, surface runoff and erosion between 1987 and 1991 are included.

Since November 1986 the present author has been active as research assistant for the project on the Swedish side and also spent altogether 14 months full time in the field at the research area until January 1993.

2.3 Objectives

The goals of this research project, included by this thesis work, have two major objectives; 1) to quantify effects on runoff, erosion and streamflow dissolved nutrient losses when clear-felling tropical rainforest with or without soil disturbance and subsequent burning of residues before the establishment of forest plantation, 2) to increase the ability of transfer of this knowledge by explaining differences in streamwater nutrient losses after different treatments from understanding of the interaction of erosion-, leaching- and transport processes. Problems of erosion on disturbed soils have been a recognized problem for a long time. However, the implications of erosion on losses of nutrients is a complex matter, not deeply investigated in the specific rainforest environment. One major hypothesis of the project was that burning of residues in this warm and humid environment increases nutrient losses, leading to significant reduction in regrowth of the planted forest. Another hypothesis was that soil disturbance by heavy tractors also contribute to nutrient losses and reduces the regrowth of the plantation.

3. RESEARCH AREA AND METHODS

3.1 Research area

The SFI watershed research area at Mendolong is situated at 650 - 750 m.a.s.l. on the foothills of Gunung Lumako (1967 m.a.s.l.) in the Crocker range (115.5°E, 5.0°N) in the state of Sabah, Malaysia (Figure 1). Mendolong is a tree nursery in the center of the forest plantation areas, 35 km southeast of Sipitang at the Brunei bay on Sabahs' west coast, close to the Sarawak border. The small village at Mendolong tree nursery includes the center for the SFI tree breeding and species suitability trials (Sim and Gan, 1988) as well as soil trials and surveys (Wong, 1989). The facilities in Mendolong include the soil and water laboratory built up within the frames of this impact study. The research area is easily accessible from Mendolong on about 3 km of gravel sealed road. Further details on the vegetation and physical environment of the research area are given below under the description of the site ecosystem.

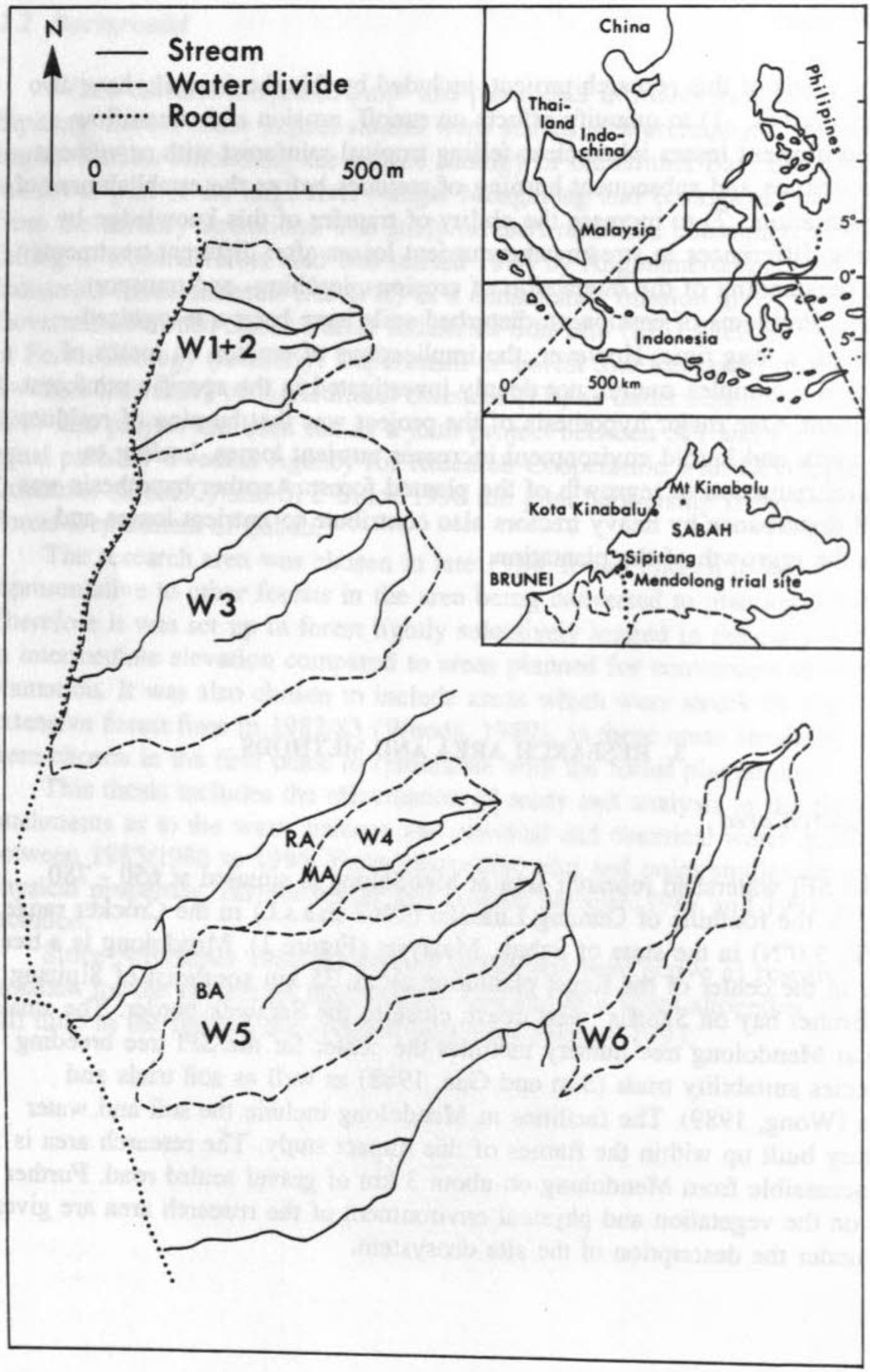

Figure 1. Location and catchments of the Mendolong research area in Sabah, Malaysia. Also runoff plots RA, MA and BA are marked (cf. paper VI).

3.2 Forestry treatments

Catchments W1 - W3 comprised secondary vegetation after the forest fire. W1 and W2 were 100% covered by the effect of the forest fire, while W3 had 80% of surface damaged by forest fire. Catchments W4 - W6 comprised tropical rainforest lightly selectively logged in 1981. The treatments, covering 100 % of the catchments, were as follows:

W1 Non-mechanized clearing of remaining trees and secondary vegetation, no wood extraction, burning of all biomass and planting.

W2 The same treatment as W1.

W3 Control catchment for forest fire area. No treatments.

W4 Manual felling, manual wood extraction, clearing of planting rows in the slash and planting in these rows, without burning.

W5 Manual felling, wood extraction using crawler tractors, burning of the remaining biomass and planting (normal practice).

W6 Control catchment for selectively logged area. No treatments.

Installation of streamflow gauges and raingauges were made in 1985. Reference monitoring was started in mid 1985. Felling and extraction of treated catchments were carried out from November 1987 to January 1988. Burning and rowing was done in February to April, and planting was conducted at W1+2 and W4 in March and W5 in May 1988. Wood extraction methods used were described in detail in paper I. All logging and silvicultural treatments in this study were preformed by one of the regular contractors and personnel involved in full scale operations in the surrounding areas.

The normal practice of burning the residual biomass before planting was used in W1+2 and W5. About one month after burning, grasses and weeds started to colonize the bare soil, and after three months most soil, except for tractor disturbed areas, was covered with vegetation. At W4 burning was avoided and all remaining biomass was cut into smaller constituents and put into rows, leaving strips of bare ground accessible for planting.

The planted tree species was *Acacia mangium* for all treated catchments. The accumulation of above ground vegetation in trees and ground vegetation in the plantations after 1.5 and 3.7 years are presented in Table 1. According to normal practice, manual circle weeding at the planted trees combined with slashing in between trees was made 4 times during the first 2 years for all 3 planted catchments. In November 1990, also manual spraying around the trees

***Table 1.** Above ground biomass dryweight accumulation (t ha^{-1}) in Acacia mangium plantation in different catchments 1.5 and 3.7 years after planting in Mendolong research area, Sabah, Malaysia (after Sim and Nykvist, 1990 and Nykvist et al., 0000). W1+2 was manual clearing of secondary vegetation after forest fire and burning before planting, W4 was clear-felling of tropical rainforest, manual extraction and no burning before planting, W5 was clear-felling of tropical rainforest, tractor extraction and burning before planting.*

catchment	biomass accumulation (t ha^{-1})			
	1.5 years		3.7 years	
	trees	undergrowth	trees	undergrowth
W1+2	2.3	6.2	15.8	6.4
W4	10.5	1.8	44.6	5.4
W5	5.4	3.3	23.0	7.4

was applied in W1+2 to stop suppression of the trees from the extensive secondary vegetation, mainly consisting of herbs (*Eupatorium oduratum*), grass (*Imperata cylindrica*) and ferns (*Neprolepis biserrata*) (Sim and Nykvist, 1991; Nykvist et al., 0000).

3.3 Research methods

Continuous streamwater gauging in the six catchment streams started in mid 1985 together with measurements of precipitation. The methods of field monitoring and data handling were described in paper III. Sampling of streamwater for determination of suspended sediment load during high and low flows started in mid 1986. Sampling and analysis of the 2187 samples taken up to mid 1989 was described in detail in paper II. The sampling of streamwater for dissolved element chemical analysis was started in November 1985. Up to December 1990, 2365 samples from high and low flows were analyzed in the field laboratory in Mendolong, Sabah and in Sweden. Further details on sampling and analysis were given in paper IV. Some sampling for analysis of rainwater chemistry was made between 1986 - 1990 (IV). The intensity of sampling for both suspended- and dissolved load analysis in all streams were intensified during and after the treatments in late 1987 and during 1988.

Areal coverage of soil disturbance after manual- and crawler tractor log extraction was surveyed. This was combined with measurements of steady state infiltrability, sorbtivity and dry soil bulk density in control forest as well as different types of disturbed soils on two soil types. Methods of investigation for these studies were given in paper I. Also field measurements of soil saturated hydraulic conductivity in control forest and in disturbed soils were carried out in late 1989 (VI, after Andersson, 1990).

Seven large unbounded runoff plots were installed in control forest, W4 and W5, right after the completion of the treatments in early 1988. These plots collected surface runoff and eroded sediments carried by surface wash. Layout, installation, sampling and analysis for this experiment was described in paper VI. Observations on rain intensity, soil moisture in slopes and gully erosion on tractor tracks were also included in this last paper.

Apart from the research covered by papers included in this thesis, references will be made to other studies included in this research project. These include studies on biomass before and after treatments (Sim and Nykvist, 1991) as well as estimations of nutrient contents in the biomass of the rainforest and the plantations combined with tree growth estimations (Sim and Nykvist, 1991 and Nykvist et al., 0000). Comparison of biomass and contents of nutrients in the boles of different rainforest ecosystems were made by Nykvist (1992).

Details on data handling and statistical analysis of different datasets are reported in the respective papers (I - VI).

4. THE SITE RAINFOREST ECOSYSTEM

4.1 Bedrock and soils

As mentioned above, the research area is situated in a hilly area on the rim of the mountain range. The topography in the range consist of steep ridges and occasional peaks, and the foothills of long and more gently sloping ridges between tributaries of the two major rivers draining the area; Sungai Mengalong and Padas. The catchment research area is situated on the slope of such a ridge and the small catchments are drained by first and second order streams close to the ridge top in a landscape with high drainage density of small streams surrounded by short slopes. Mean drainage density of the catchments in this study were 63.6 m ha^{-1}. The streams of the research area are part of the Sungai Mengalong watershed. The bedrock of the research area belongs to the Maligan sedimentary formations of Miocene age (Wong, 1989 after Liechti, 1960). Sandstones and siltstones are most common but interbeded shales occur within the research area (IV). Areal distribution of the sandstone and the shale as parent material for soil formation have not been possible to map in the catchments, as the saprolite is uncovered in few places and due to probable high variability from the interbedded nature and steep dipping of the bedrock. However, bedload material and streamwater chemistry indicate a higher occurrence of shales in W3 compared to other catchments (IV). The shale was richer in nutrients compared to the sandstone, especially for S and Mg (IV).

The soils of the research area consists of two main types, namely Orthic Acrisol and Gleyic Podsol. The Podsol areas were located in more gently sloping areas with loamy sand top soils while the Acrisols had higher clay contents in top soils (IV). Some soil physical properties of the two soils were reported in papers I and VI. The uppermost 20 cm of the soils were loose with well developed structure and high porosity. These properties were most pronounced in the clay topsoil with porosities up to 67 % giving higher steady state infiltrability than the loamy sand. The structure of the loamy sand top soil was also more shallow than in the clay soil as indicated by hydraulic conductivity which showed a steeper decrease with depth than in the clay soil. A positive relation between the degree of decomposition (organic C/N ratio) of dissolved organic material in streamwater and the catchment areal percent of Podsol was showed during the reference period before treatments (IV). However, this first 2.5 years of study were relatively dry, and later during shorter and wet periods also streamwater from W3 (100 areal percent clay topsoil) showed high C/N ratios.

4.2 Vegetation and biomass

The natural vegetation of the research area is a lowland hill dipterocarp forest (Whitmore, 1984), although the altitude is just below the transition to lower montane forest. In 1981 the forest of the research area was lightly selectively logged. Sim and Nykvist (1991) presented data from forest inventory and a biomass estimation of the catchments W4 and W5. They also presented the contents of macro nutrients of different compartments of the biomass, but here only a brief description of vegetation and biomass will be extracted. Number of tree species among the harvested trees with girth above 60 cm were more than 50. Number of trees per hectare were 146 and total above ground biomass were 261 t ha^{-1} of which 4.2 t ha^{-1} was located in other plants than trees. This figure falls within the lowest range of natural tropical rainforests from comparable studies, due to the selective logging. Notable is however that the biomass of leaves and leaf area index (6.7) was comparable with the forest of the Pasoh forest of Peninsular Malaysia (LAI 6.9) with almost double above ground biomass. Below ground biomass (including tree stumps) were 179 t ha^{-1}, of which the biomass of small roots (< 20 mm) of 26 t ha^{-1} were strongly concentrated to the upper 20 cm of the soil.

The secondary vegetation after the forest fire of 1982/83 was also used as a control in this research project. Sim and Nykvist (1991) also reported on biomass estimations of this secondary vegetation in catchment W1+2 made in 1985 and 1988. Above ground vegetation in 1985 was 4.7 t ha^{-1} dominated by ferns, grasses and herbs (mainly the same three pioneer species as was mentioned above (3.2) as weeds in the plantation). In 1988 total above ground biomass had increased to 26.7 t ha^{-1} of which 20.4 t ha^{-1} in small trees.

4.3 Climate and water balance

The climate of southwestern Sabah could be described as tropical humid climate with a fast decreasing maritime and monsoonal influence with distance from the coast. Mean monthly temperatures are typically between 17° - 30° C (Wong, 1989), with moderate seasonal variation and the highest temperatures experienced at the coast on low elevations and the lower temperatures at higher altitudes in the mountain range. At Mendolong tree nursery (500 m.a.s.l.), in open plantation, monthly minimum temperatures typically range between 20-22° C and maximum temperatures between 27-31° C (1987, SFI, 1988). Under rainforest in the research area, minimum temperatures have been observed to be similar but maximum temperatures lower compared to the tree nursery in the plantation area (unpublished). The monsoons give rainfall peaks between equinoxes, most apparent close to the coast. Convective rains on the other hand occurs throughout the year, but could be expected to be higher during the equinoxes with maximum insolation. In the Maligan valley, behind Gunung Lumako and the southernmost part of the Crocker range, no monsoonal influence can be traced and annual rainfalls are much lower than elsewhere in this part of the state (Wong, 1989). Also indications of local rainfall maxima on the slopes of mountain ridges and peaks in the mountain range along the coast caused by orographic uplift have been reported (III). Walsh (1982) discussed similar effects at nearby Gunung Mulu National Park in northeastern Sarawak.

A five year mean water balance for the two control catchments in the research area can be seen in Table 2 (after III). The five years presented were the hydrologic years of 1985/86 - 1989/90. The hydrologic year is chosen from August to July as the driest periods in the research area occurred during June and July.

The small scale areal variation of rainfall was high in the research area, with almost 1000 mm maximum annual difference between different raingauges (III). That underlines the importance of convective rain for the research area, giving high areal variability in rainfall. Periods with more than 10 days without rain occurred regularly around July and December (III). The most wet year of 1988 experienced 190 rainy days, but 55 of these resulted in less than 8 mm of rain and only 21 days in more than 50 mm rain (VI).

Table 2. Mean water balances of control catchments in Mendolong research area, Sabah, Malaysia for hydrologic years 1985/86 -1989/90. Catchment W3 was Orthic Acrisol with clay top soil with secondary vegetation after forest fire 1982/83 and W6 was Gleyic Podsol with loamy sand top soil with lightly selectively logged rainforest.

catchment	precipitation (mm)	runoff (mm)	evapotranspiration (mm)
W3	3215	1962	1253
W6	3490	1950	1540

Mean catchment derived evapotranspiration of the forest in W6 was well in accordance with earlier studies of lowland tropical rainforests (III). Bruijnzeel (1990) made a critical review on reported evapotranspirations from 15 studies in lowland tropical rainforests. He calculated an average of the most reliable 6 - 11 studies to 1400 - 1430 mm per year, rather independent of different yearly rainfall at different sites. However, he pointed out that annual evapotranspiration for a certain rainforest site tends to be correlated to annual rainfall, as annual interception may increase in wet years (eg. Blackie, 1979; Shuttleworth, 1988) and transpiration can be limited by soil moisture deficits during dry years (eg. Baillie, 1976; Dietrich et al., 1982). That relation between yearly evapotranspiration and rainfall is also confirmed from the Mendolong catchments (Figure 2).

The ecological significance of dry periods in the region was discussed in paper III. Baille (1976) concluded them not to be a problem in Sarawak, with the exception of shallow soils with low storage capacity. Large trees have been reported to have considerable rooting depth (eg. Baille and Mamit, 1983) and water has been shown to be extracted from at least 1.5 m depth (eg. Calder et al., 1986), but seedlings and planted trees probably experience moisture stress periodically (III and tensiometer data in VI).

4.4 Slope hydrology

The flow path of water from the delivery to the soil surface to the outflow in the stream is decisive for the transport of eroded sediments (Douglas and Spencer, 1985) and the leaching, transport and losses of solutes from slopes (Pilgrim et al., 1979; Burt, 1986). Since Hortons early work (1933), separating between infiltrating water and infiltration excess overland flow, there has been many models proposed to explain water movements from rainfall to generation

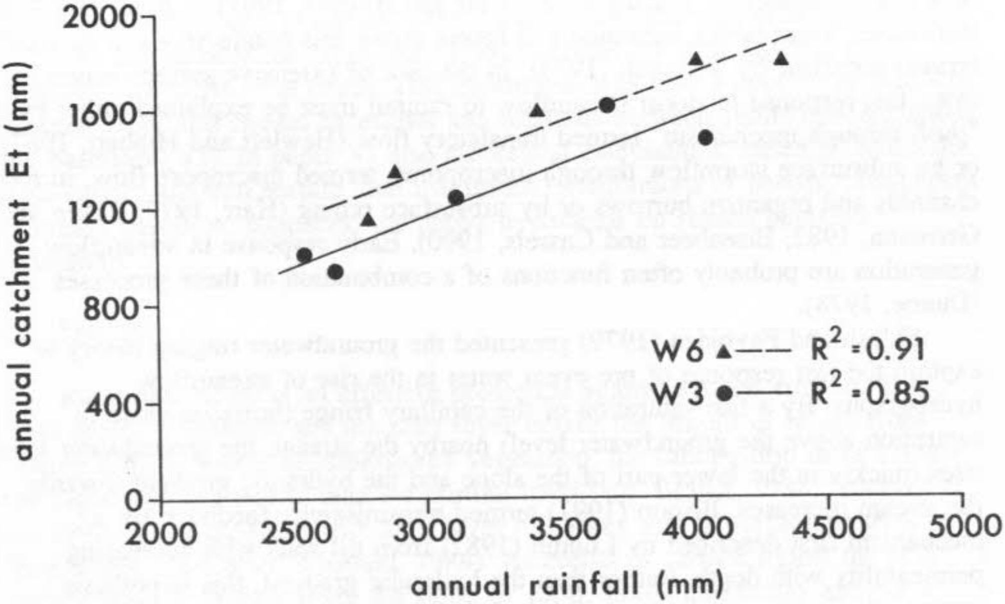

***Figure 2.** Annual catchment derived evapotranspiration (mm) as a function of annual rainfall (mm) for control catchments W3 and W6 in Mendolong, Sabah, Malaysia. W3 were 2 - 7 years old secondary vegetation after forest fire and W6 were lowland dipterocarp forest lightly selectively logged 5 - 10 years earlier.*

of streamflow in different environments. The variability in areal delivery of rainwater to the rainforest floor, as well as permeability and macropore distribution, can be very high in the rainforest environment (eg. Lloyd and Marques-Filho, 1988; Elsenbeer and Cassel, 1990). Even very high infiltrability may be exceeded in points with concentration of water for example by stemflow at the base of tree trunks (Herwitz, 1986). Such locally derived infiltration excess overland flow may be infiltrated in other parts of the slope. Net surface runoff for whole slopes will be highly dependant on means of infiltrability and hydraulic conductivity at different horizons in relation to soil saturation. This is clearly demonstrated by earlier studies on runoff from different soil horizons in different tropical rainforest pedological environments (Douglas and Spencer, 1985).

Bruijnzeel (1990) and Anderson and Spencer (1991) reviewed research on tropical rainforest slope hydrological processes and Bishop (1991) made a current review on theories of runoff generation. In the humid tropical situation, throughflow (lateral flow in the soil profile) (Kirkby and Chorley, 1967) in shallow soil horizons has shown to be an important path for the bulk of water from the soil surface to the stream. The throughflow can occur as unsaturated flow called subsurface stormflow (Hewlett and Hibbert, 1967) or more rapidly in perched water tables, saturated soil layers above less conductive soil layers, termed interflow by Weyman (1973). In the lack of extensive surface runoff, an often fast response in storm streamflow to rainfall must be explained either by a "push through mechanism" termed translatory flow (Hewlett and Hibbert, 1967) or by subsurface stormflow through macropores, termed macropore flow, in root channels and organism burrows or by subsurface piping (Harr, 1977; Beven and Germann, 1982; Elsenbeer and Cassels, 1990). Early response in streamflow generation are probably often functions of a combination of these processes (Dunne, 1978).

Sklash and Favolden (1979) presented the groundwater ridging theory to explain the fast response of pre event water in the rise of streamflow hydrographs. By a fast saturation of the capillary fringe (horizons close to saturation above the groundwater level) nearby the stream, the groundwater level rises quickly in the lower part of the slope and the hydraulic gradient towards the stream increases. Bishop (1991) termed transmissivity feedback for a mechanism first described by Lundin (1982) from till soils with decreasing permeability with depth. Rather than the hydraulic gradient, this hypothesis stresses the increase in transmissivity of shallow soil layers with increasing hydraulic conductivity as a response to a little increase in soil moisture saturating larger and more conductive soil horizons. In the case of saturation at the soil surface, for example with surficial ground water level in bottom of concave slopes and in wide valley bottoms, saturation overland flow and returnflow of groundwater to the surface may occur (Dunne and Black, 1970; Dunne, 1978).

To understand site slope solute removal and mechanisms for nutrient leaching and erosion it is essential to have a clear picture of streamflow generation processes also including the relative contribution of pre event water of different transit time (Northcliff and Thornes 1989; Bishop 1991).

Plot investigations on the clay soil of the research area revealed only 2.9 % of rainfall to occur as surface runoff in the very slopes, restricted to high magnitude storms (VI). These events were concluded to be caused by infiltration excess overland flow during the most intensive parts of storms. Thus steepness and length of slope was determinant for accumulated volumes of surface runoff in the slopes. Because of the low occurrence of slope surface runoff together with strongly decreasing hydraulic conductivity, translatory flow and throughflow in the shallow, highly permeable top soil was concluded to be the main path for water along the slope to the stream during stream stormflows (VI). Also saturated overland flow and returnflow on the concave bottom of the slope adjacent to the stream was concluded to be likely to support stream stormflow.

Almost continuous deviations from drainage equilibrium (DDE, VI) of soil suctions between the 20 and 50 cm horizons, being of the same order as the lateral gradient in total potential, were interpreted as that lateral drainage was common also in between rains, during stream baseflow as a result of low hydraulic conductivity at depth. This hypothesis was also supported by relations between DDE and stream baseflows and relations between solute concentrations and baseflow (VI). In paper VI also theoretical calculations, based on hydraulic conductivity and soil texture, and maximum contributions of deeper throughflow to stream baseflow, supported the above hypothesis on stream baseflow generation.

4.5 Erosion

Rainforest cover is an effective protection against surface erosion. The multi storey vegetation and the litter layer brakes the impact of splash from canopy drip. Without the understorey vegetation, the canopy drip in fact often has higher erosive power than the rainfall itself (e.g. Brandt, 1988; Nortcliff et al., 1990). Here especially the litter layer has a key role in surface protection (Bruijnzeel, 1990 after Wiersum, 1984). At events of surface runoff the effect of surface wash is broken along the slope by roots, fallen tree trunks and other obstacles diverging flows and acting as barriers for soil movement (Spencer et al., 1990). On the other hand concentration of flows in the same manner could also lead to rill erosion.

All forms of surface erosion will be accelerated with natural disturbances of vegetation and litter cover on top soils. Such disturbances will occur at more rare high magnitude storms or tectonic activities (eg. Pain and Bowler, 1973), initiating tree falls and landslides, or by fire and lighter human interference like shifting cultivation (eg. Lovejoy et al., 1983). Sinun et al. (1992) also discussed the role of soil faunal activity as a source of bare ground susceptible to splash erosion. Gully erosion has been reported under rainforest cover, but only at more extreme events and then initiated by treefalls, landslides or collapsing subsurface pipes (Ruxton, 1967; Turvey, 1974; Morgan, 1986). Once formed, the erosion in a gully is less dependant on protecting vegetation cover (Bruijnzeel, 1990). In the view of continuous natural disturbances, the rainforest can be described as a highly dynamic system of relations between denudating processes, treefalls and vegetation colonization and successions (Spencer et al., 1990), rather than older views of a stable and fixed ecosystem (eg. Richards, 1952, 1969).

Consequently, depending on time and magnitude of storm events, also the sediment delivery to the stream will be very variable. Also in the stream there are temporary storages combined with streambank erosion (Spencer et al., 1990). Estimations of sediment delivery ratio between sediment supply by erosion and catchment sediment yield in streams will depend on how, where and when measurements of input and output are made (eg Walling, 1988; Bruijnzeel, 1990).

The rate of all erosion processes described above, and especially basin sediment yield, are highly dependant on the time perspective. In a geomorphological perspective the rare high magnitude events play a highly determinating role for total soil loss. Also climatological and geological factors make comparison between different sites difficult.

In this study recorded slope surface erosion of 0.04 t ha^{-1} under forest cover (VI) was low compared with reviews on surface erosion from undisturbed (sub)tropical forests (Wiersum, 1984). Measurements were made during the wettest year out of five (VI). Only some splash erosion was observed under forest cover. As a consequence of low occurrence of slope surface runoff, no sediments stored in the slope in traps like roots or larger debris were activated by rill or surface wash. As the plots under forest cover did not include any surfaces of natural disturbances discussed above, the recorded surface erosion rate was below that of the catchment stream suspended sediment yield (III). Catchment suspended sediment yield under forest cover (W6) was 0.25 t ha^{-1} and from the secondary vegetation control (W3) 0.42 t ha^{-1} (II), which was also low compared to other studies (Lal, 1986; Douglas and Spencer, 1985; Douglas et al, 1992).

4.6 Slope solute removal

As the rainwater interact with the biosphere and the pedosphere on its way to the stream it will obtain different quality as to dissolved elements depending on time and the physical, chemical and biological environments of its pathways. Exchange and weathering on surfaces of soil particles and weathering of bedrock will supply ions in solution to soil water. Naturally the relative contribution of elements from weathering will be the highest to water in small pores with larger particle surface per volume. Consequently water of longer contact and transit times, travelling in smaller pores, will gain higher concentration of ions of geochemical origin (Pilgrim et al., 1979; Burt, 1986). Release of elements from the biosphere to the water will occur by leaching from living tissue, and by leaching of elements from the mineralization of litter and other dead tissue, as well as of humus at or below the soil surface (Gosz et al., 1973). In cold and temperate climates concentrations of throughfall has been shown to be lower than that of runoff in litter and throughflow below the humus layer (Cole and Johnson, 1977; Foster and Grieve, 1984). The same relation has been reported from tropical rainforests (eg. Sinun et al., 1992). Biological processes also reduces concentrations, as microbial fauna and flora as well as higher plants extract nutrients from the soil solution.

Between the water and these mineralogical and biological sources of elements, physical, chemical and biological processes of precipitation/dissolution and sorption/exchange take place. Because of the larger ratio of surface to volume and longer periods of water contact, these processes will occur mainly on surfaces of organic- or mineralogical colloidal particles or small roots in the smaller pores of the soil. The speed of the processes and the residence time of water will determine to what extent processes approach equilibrium (Bache, 1990). Several studies have discussed physical and chemical nonequilibrium to be prevalent in soil water throughflow during stormevents (eg. Birkeland, 1974; Beven and German, 1982) by soil water bypassing micropores. Hydrochemical processes controlling transport through various pore classes were addressed by Wilson et al. (1991). They explained flushing of elements in storm throuhflow with mixing of new water entering mesopores with solute of micropores. They also proposed increased soil air pressure to increase mixing by exfiltration of micropore solute into the mesopores. However, that process would be less effective in wetter soils. In this study, under forest cover, mesopores (1 - 0.01 mm, Luxmoore, 1981) at 20 cm depth were normally not saturated between rains (>0.3 kPa) but never completely drained (<30 kPa), with soil suction having inter-rain weekly means of less than 4 kPa in wet periods and seldom above 10 kPa in dry periods (VI).

In the humid tropical environment, the importance of a bi-phasic flow regime of bulk throughflow with short transit time trough large pores has been stressed as an important nutrient conserving mechanism by Northcliff and Thornes (1978 and 1989) and Sollins and Radulovich (1988). A schematic picture of the relationship of slope flowpaths and solute concentrations is given in Figure 3 (after Burt, 1986).

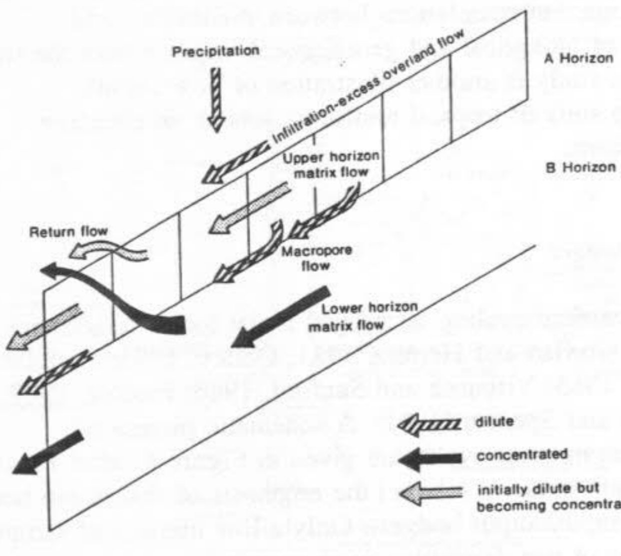

***Figure 3.** Relationship between type of hillslope runoff and its solute concentration. After T.P. Burt, 1986.*

Like for the case of erosion (4.5 above), it can be relevant to discuss the effect of episodic natural disturbances on soil water and streamwater dissolved concentrations and losses. Bruijnzeel (1990) made a small review on the subject. Soil water from small gaps have not shown differences in that aspect compared to surrounding forest (Parker, 1985; Uhl et al., 1988). However, already small cleared plots of 500 m^2 showed similar rises in concentrations as was leached from larger plots (Parker, 1985). The critical area of disturbance to create increase in soil water concentrations would firstly be dependant on the amount of elements released depending on amount of decomposing biomass and the rate of its mineralization, secondly from reduction in biological uptake and exchange by area of bare soil, damage to the root mat and decreases in active biomass as well as increased runoff from loss of interception and transpiration. Larger natural disturbances as landslides could theoretically be expected to create increased dissolved losses like the case of clear-felling and soil disturbance (V and below).

A discussion of relations between slope hydrology of the research area and the streamwater chemistry were given in paper VI. Differences in concentrations of elements of biological and geochemical origin in different baseflows (IV) could be related to hypothetical differences in sources for runoff during runoff recession. In this study 70 - 74 % of runoff in control catchments occurred in stream stormflow (III), and the bulk of that water was concluded to move as shallow throughflow concentrated to larger pores (VI). Also large parts of baseflow was concluded to spring from water of similar sources. Apparently there were no differences between mean stream stormflow concentrations and mean baseflow concentrations, but correlations between streamflow and concentrations of elements of biological and geochemical origin within the range of baseflow (IV). Thus, this study is another illustration of how runoff generation in permeable top soils of tropical rainforest acts as an effective nutrient conserving mechanism.

4.7 Nutrient input/output budget

Extensive reviews of nutrient cycling in tropical moist forest ecosystems have been given elsewhere (Jordan and Herrera, 1981; Golley, 1983; Douglas and Spencer, 1985; Jordan, 1985; Vitousek and Sanford, 1986; Proctor, 1987; Bruijnzeel, 1990; Anderson and Spencer, 1991). A schematic picture of transports and pools in forest nutrient cycles are given in Figure 4 (after Proctor, 1987). Except for slope solute removal (above) the emphasis of discussion here will be restricted to simple input/output budgets. Only a low number of samples of rain for chemical analysis of wet deposition were sampled, compared to the effort in sampling of streamwater, to relate the chemical composition and (lack of) seasonal variability to other studies in the tropics and the region (IV).

The problems of estimation of atmospheric nutrient input were currently reviewed by Bruijnzeel (1990) and Rosén (1990). Conventionally, atmospheric input in studies of nutrient budgets for tropical forests have been approximated with bulk precipitation. That may be seen as an integration of wet and dry deposition, at least for the collector itself (Bruijnzeel, 1990). In this study samples for rain chemistry were collected by mounting the collector at the start of the rain, meaning that dry deposition in the collector in between rains were not included, making the sample more strictly a wet deposition sample. Therefore concentrations were low and problems of contamination of samples reduced. Samples of bulk precipitation are difficult to interpret as the relation of particle capture between the collector and a forest is hard to estimate. Also some particles like pollen does not in many cases represent an input but a mere circulation within the system.

Solute concentrations of nutrients in rainwater depend on proximity to the sea, but are generally very low compared to nutrient concentrations of water in other compartments in the nutrient cycle (eg. Eriksson, 1960; Bruijnzeel, 1989; Veneklaas, 1990). The problem of adequate sampling of wet deposition is clearly shown in the high variability of nutrient input of 25 (sub)tropical sites reviewed by Bruijnzeel (1990). Risk for contamination of the dilute rain water

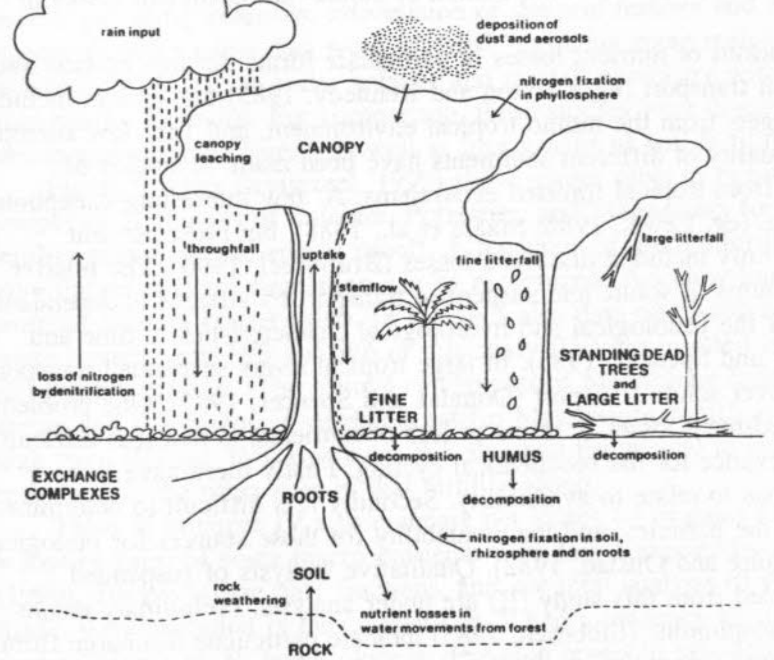

***Figure 4.** Tropical rainforest nutrient cycles. After J. Proctor, 1987.*

sample is high, whereas high standards of sampling and sample handling must be maintained. In the humid tropical environment also a seasonality in the marine influence can be expected under monsoonal influence (eg. Manokaran, 1980). As the precipitation in the research area was dominated by convective rains, concentrations of rainwater was low compared with literature (IV).

Nutrient discharge in budgets of forest ecosystems are assessed by analysis of soil water sampled by lysimetry and in throughflow troughs below the root zone, or streamwater draining catchments of different sizes and maybe including several different ecosystems with or without human interference. The interpretation and comparison of different studies is therefore not a simple matter (Bruijnzeel, 1990). The quantification of the export from soil water sampling demands estimations of drainage, for example obtained by different degrees of insecure assumptions of the water balance. Furthermore in slope hydrological situations as described above large analytical errors will be obtained from only analyzing soil water of deeper horizons, especially when quantifying effects of treatments (IV, V and VI). In that sense the catchment approach is more secure, provided that the catchment is watertight and is homogenous as for type of ecosystem and land use. Small catchments are more sensitive to errors in measurements and large ones will have a problem with homogeneity. In the sense of representative sampling of water leaving the ecosystem, also the adequate sampling of both storm- and baseflows are essential in the catchment approach (IV and V). Reynolds et al. (1990) made a current review of methods for computation of solute- and sediment losses in streams.

Quantifications of nutrient losses in particulate forms through erosion and stream sediment transport (eg. Clayton and Kennedy, 1985) are seldom included in nutrient budgets from the humid tropical environment, and very few attempts to access the quality of different sediments have been made in studies of nutrient losses from tropical forested ecosystems. A few interesting exceptions have been made (eg. Lewis, 1986; Maass et al., 1988), but most nutrient balances made only included dissolved losses (Bruijnzeel, 1990). The relative division of amounts of solute and suspended transport by streams is dependant on variations of the pedological and hydrological characteristics in time and space (Douglas and Spencer, 1985). In large tropical rivers suspended transport is dominating over solute transport (Douglas and Spencer, 1985). One problem in evaluating sediment losses in terms of loss of nutrients, is that it is difficult to relate to its relevance for the biochemical cycling. Firstly there have to be a choice of analysis to relate to availability. Secondly it is difficult to determine the sources for the particles, and the availability for those sources for biological cycling (eg. Young and Onstad, 1982). Qualitative analysis of suspended sediments sampled from this study (II) are under analysis. Preliminary results for available Phosphorous (Sibbesen, 1983) indicate particulate discharge from control catchments (unpublished) of the same order as by dissolved discharge (about 0.1 kg ha^{-1} y^{-1}, IV). Control catchment suspended sediment yield under forest cover (W6) was 0.22 t ha^{-1} y^{-1} and from the secondary vegetation control (W3) 0.48 t ha^{-1} y^{-1} compared to total dissolved load of 0.16 and 0.25 t ha^{-1} y^{-1} for the same catchments respectively (IV). However, included in the data presented in this thesis only included dissolved losses, in accordance with earlier studies.

Neither gaseous losses are normally included in nutrient balances of forest ecosystems. Mainly sulphur and nitrogen compounds could be considered for gaseous losses. In special environments and circumstances these losses may be considerable, for nitrogen at oxygen deficiency and high pH as well as by fire. H_2S might be lost from plants, especially in sulphur rich soils (Winner et al., 1981). In this study there were high loads of dissolved sulphur from catchment W3, concluded to depend on high contents of sulphur in shales locally interbedded in the bedrock (IV). However, significant gaseous losses in comparison with streamwater losses were not considered likely from control catchments in this study. Consequently, and in accordance with other studies, gaseous losses were not included in the nutrient budgets presented in paper IV.

Several reviews on input/output budgets of tropical rainforest ecosystems have been made (cf. nutrient cycling, top of this paragraph). These comparisons have often been falling on difficulties in comparison of different methodologies, type of analysis and laboratory standards (eg. Vitousek and Sanford, 1986). In that view, Bruijnzeel (1990) made a comprehensive review on a larger data set (n=25), including extensive information of site soil fertility and methodological aspects. In some cases data were adjusted "to obtain more realistic results" by replacement of data on water balance and rainfall chemistry with "better" data from nearby locations. For comparison, these studies were divided into lowland forests and montane forests as well as four groups of soil fertility.

Figure 5 (after Bruijnzeel, 1990) show scatter plots of nutrient yields (output or discharge) of Calcium, Potassium and Magnesium for different soil fertility groups versus annual runoff. A tendency of more infertile systems to have tighter nutrient circulation can be recognized. Also low humidity seems to promote nutrient conservation. In Figure 5 also data on dissolved discharge from control catchments W3 and W6 of this study (IV) are inserted in their respective soil fertility classes. The control catchments of Mendolong falls well within the scatter and the ranges of yields except for Magnesium, which is comparably high from W3 (cf. IV). However, W3 also had the second to highest runoff compared to the other sites in that fertility group.

Table V in paper IV also compare net losses (-) or gains (+) (input - output = loss or gain) of some macro nutrients with data compiled by Bruijnzeel (1990). Studies where only fractions, and not total analysis of P and N were made, were excluded in the comparison. P had a tendency to accumulate in most studies. However, P- levels close to detection levels in rain and streamflow make any conclusion insecure (IV). In this study both control catchments were

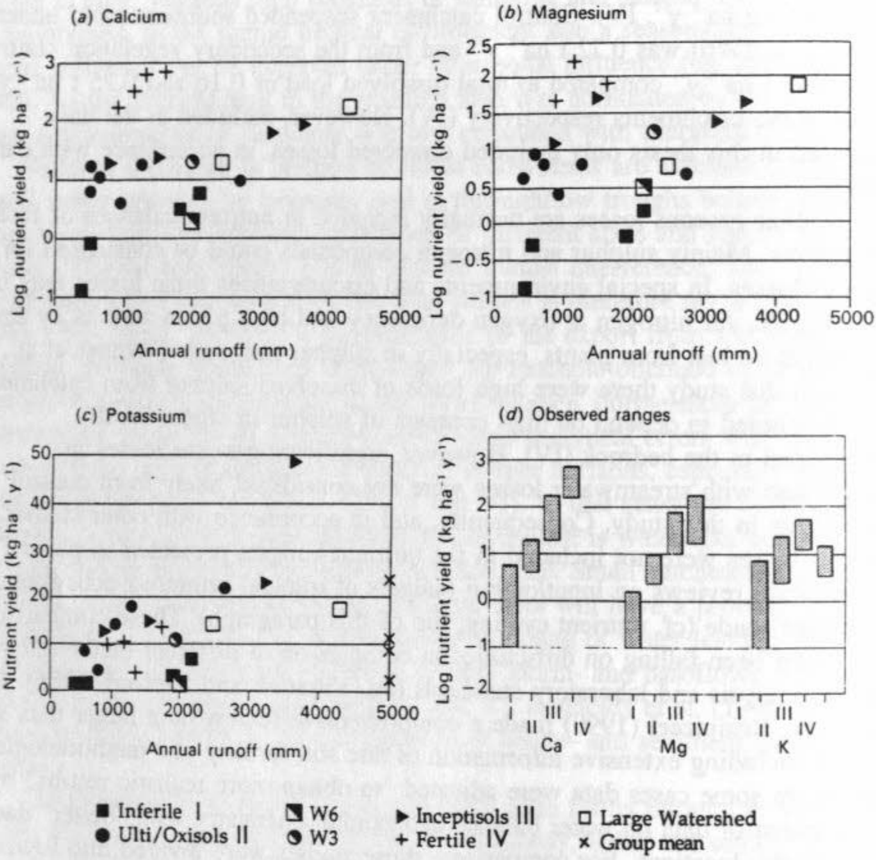

Figure 5. Scatter plots of annual runoff versus nutrient yield (discharge) for 20 selected (sub) tropical forest ecosystems. (a) Calcium, (b) Magnesium,(c) Potassium, (d) observed ranges. After L.A. Bruijnzeel, 1990 and 1991. Control catchments W6 (infertile) and W3 (Ultisol/Oxisol) from mendolong are included.

found to loose P. A more secure conclusion was that streamwater output were larger from W3 with secondary vegetation and more fertile soil. As N in the biogeochemical cycle includes several processes including gaseous fixation and losses, the water based input/output budgets for this element become even more insecure. For example from the oxisol at San Carlos, Venezuela 58 % of N input was derived from biological fixation, and denitrification was 17 % of total output (Jordan et al., 1982). In this study W3 had a larger gain of N than W6 due to larger streamwater discharge from W6. That difference in accumulation might be attributed to the secondary growth succession in W3 (IV). Paper IV also discussed the possibilities of worse conditions for mineralization in the Podsol as indicated by higher dissolved organic C/N ratio and hydrologic conditions.

The most clear difference between the two catchments of different soil types, apart from the dominance of organic Carbon (mostly organic acids, IV) and low pH of the streamwater from the Gleyic Podsol of W6, were higher gross and net losses of Sulphur, Silica and major cations from the Orthic Acrisol of W3. The larger discharge and loss from W3 were concluded to be an effect of difference in the relations between soil type, weathering and bedrock of W3 (IV). Differences in efficiency of nutrient conservation between the forest of W6 and the secondary succession after the forest fire in W3 was also discussed as a possible reason, but as the large effects from the clear-felling and burning treatments were rather short lived, soil and bedrock were considered more important during the time of monitoring, which started almost three years after the forest fire.

Input/output and loss or gain of all elements included in the study are presented in Table IV in paper IV. An interesting remark can be made on the apparent accumulation of Chloride in both catchments. Conversely, at treatments Chloride were released in large amounts into streamwater (V). Remarks on similar conditions in Swedish investigations were made by Rosén (1990).

5. TREATMENT EFFECTS ON DYNAMICS BETWEEN HYDROLOGY AND NUTRIENT LOSSES

5.1 Harvest

5.1.1 clear-felling

The effect of clear-felling on the water balance was discussed in paper III. The loss of transpiration and interception leads to increase in water yield (Bosch and Hewlett, 1982). Bruijnzeel (1990) reviewed the hydrology of tropical forests and effects of conversions. Still very few studies have been made to quantify increase in water yield due to different conversions of tropical rainforest (III). The other major effect considered here were the loss of nutrients in streamwater induced by the decomposition of dead biomass after the clear-felling (V) together with surficial drainage (VI). Bruijnzeel (1990) highlighted the scarcity of information on hydrological losses of nutrients from early stages of plantation establishment at conversions from tropical rainforest. Only two such studies are presented so far. Firstly from Jari Florestal, Brazil (Russel, 1983), a study based on lysimetry and false "time series". Secondly a study in Sungai Tekam, Peninsular Malaysia, reported on the conversion of forest to cocoa and oil palm plantation (DID, 1989). The later study was based on streamwater sampling but included to few stormflow samples to reflect the flushing of elements of overland flow (Zulkifli Yusop and Abdul Rahim, 1991). Those studies showed increased nutrient concentrations in draining water, but this study reported much higher losses as 73 - 96 % of treatment induced losses were carried in stream stormflows (V). It was interesting to see that the shallow throughflow working as a nutrient conserving mechanism under forest cover, in fact can be seen as the mechanism contributing to the large losses of dissolved nutrients from decomposing slash after clear-felling (VI). Effects on water yield and discharge dynamics in the different catchments during harvest and planting were reported in paper III, and losses of nutrient dissolved in streamwater in paper V.

Streamwater concentrations of major plant nutrients had a clear response during the period of harvest as well as electrical conductivity (V). Mean monthly baseflow concentrations were raised already during November, but the largest rise were during December (Figure 6). Figure 6 show streamwater baseflow concentrations of Potassium because that element is easily leached and give a good picture of the trend for other elements (cf. V). In fact baseflow concentration of Potassium had a distinctive rise already at the 24th of November, less than 10 days after the start of the felling in the upper parts of W4 and W5. The two major rain events of 2nd and 13/14th of December are clearly reflected in streamwater recession baseflow concentrations. W1+2 had

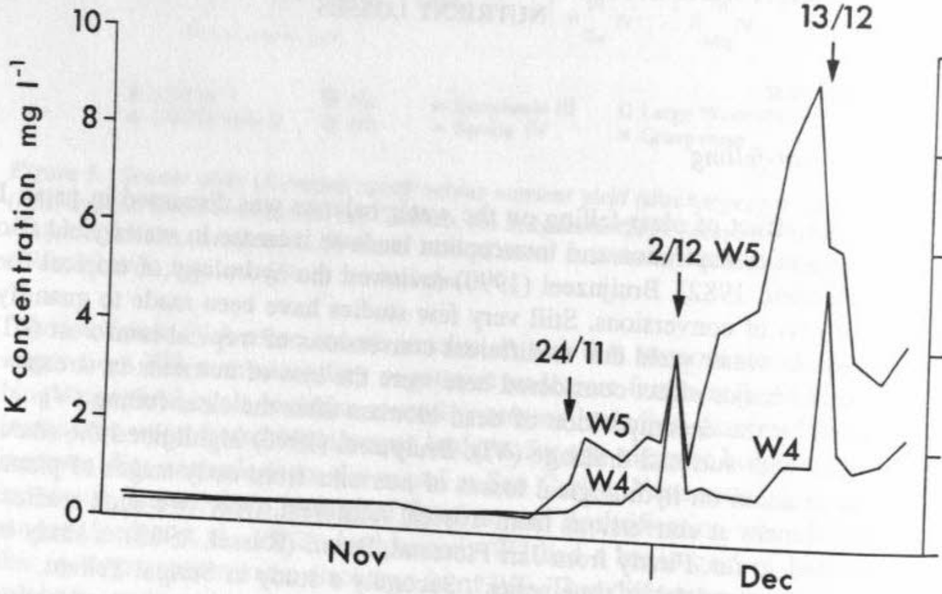

Figure 6. Dissolved Potassium concentration (mg l^{-1}) in stream baseflow from catchments W4 and W5 in Mendolong, Sabah, Malaysia during the first month of clear-felling and extraction, which started on the 16 th of November 1987. Major rainevents are marked with dates. W4 were manual extraction and W5 crawler tractor extraction.

similar rises in concentrations as W4 but later, as underbrushing of secondary vegetation started in mid December. Higher concentration levels, in accordance with those of W5 during December, were seen in W1+2 again in mid January when all small trees were felled. It is interesting to see the much higher concentrations of W5 stream baseflow compared to W4 (Figure 6), as the felling in the two catchments were advancing in about the same speed. The only difference was that during this time in December the tractor soil disturbance in W5 was as most intensive. There is no reason to believe that mineralization and leaching in the slash itself from the felled trees in W5 should be faster or more effective. However, almost all disturbed surfaces were devoid of litter, humus layer and the aggregated top soil. One explanation may be that this soil organic matter together with leaves, branches and bark from slash were mechanically grinded by the tractors on the tracks and also pushed to the sides of the track, mixed together with mineral soil. These masses in heaps along the tracks could have been favorable substrates for microbial mineralization already in the earliest stage. Also the lost top soil on disturbed surfaces contributed to lowered nutrient retention. The effect on streamwater concentration during stormflow was even more dramatic (V). During this period of harvesting, however, there were only minor differences between stormflow concentrations of W4 and W5 (IV, app. 1). The reason for the high concentration levels in W1+2 when felling the small trees in mid January may be explained by the large amount of leaf biomass cleared in very short time (two days). By comparison the felling in W4 and W5 were carried out during two months.

Actual treatment induced streamwater net losses from the catchments were presented in paper V. During harvest these losses were very much reflected by the circumstances discussed above and in paper V. For W4 the highest rate of nutrient loss was during the harvest and some four to five months there after (cf. Figure 6 in V). The loss from W1+2 was less during this period in consequence with the lower amounts of biomass in the slash (V and Sim and Nykvist, 1991). W5 had the highest loss during harvest compared to the other catchments in consequence with above and paper V.

5.1.2 soil disturbance

Use of heavy machinery is known to increase bulk density of top soils either by compaction (eg. Seubert et al., 1977) or by excavation of top soil to deeper soil horizons with higher bulk density (Van der Weert, 1974; I). The compaction consists of the collapse and compaction of aggregated top soil structure (Van der Weert, 1974) and compaction in the soil matrix (eg. Bodman and Constantin, 1965). Many factors affect the susceptibility of soils to compaction, like soil moisture, texture and structure, organic content and soil fauna, amount and spatial distribution of roots, etc. (eg. Greacen and Sands, 1980).

The increase of bulk density in the surface and top soil leads to constraints of root penetrability (Van der Weert, 1974) and decrease in soil permeability for infiltration and percolation (eg. Lal and Cummings, 1979) as well as for gas diffusion. Reduced infiltration can further lead to surface runoff and surface and gully erosion along disturbed surfaces (eg. Couper et al, 1981; Lal, 1981 and 1986) and risks for increases in nutrient leaching (Kang and Lal, 1981). Further more, the removal of top soil leads to site depletion of organic material and thereby a loss of exchange capacity (Sanchez, 1976) as well as an important substrate and source of nutrients and energy for soil faunal activity and mineralization (Jordan, 1991) and consequently worse circumstances for rehabilitation of soil structure in disturbed top soils (VI).

The effect on soil disturbance of manual felling for selective logging or clear-felling for plantation establishment using chain saws is not reported from the tropical forest. However, compared to the degree of soil disturbance induced by mechanized felling using crawler tractors with shear blades or tree pusher/root rake (Lal, 1981) and any form of mechanized log extraction method it can be considered of minor importance. For log extraction cable yarding systems can reduce the induced soil disturbance, but are in most cases not suitable for selective logging. However, even cable systems need roads for support and Greer et al. (1990) reported severe stream siltation after roading in connection with cable yard logging in Eastern Sabah. Hamilton and King (1983) and Bruijnzeel (1990) made extensive reviews of hydrologic and soil responses to uses and conversions of different intensity of disturbance. Paper I also gives an introduction to earlier studies of soil disturbances of moderate to high intensity on tropical rainforest soils.

Megahan and Sweithelm (1983) presented a summary of 16 studies in USA and Canada on areal percentage of disturbed soils caused by use of different logging systems (after Megahan, 1981), values ranging from about 35 % for tractor logging to just a few per cent for aerial logging. They, like many others, also pointed out that most hydrologic and soil effects are concentrated to road construction and especially at stream crossings, which consequently can lead to great reduction in undesirable effects with proper supervision of the planning of roading and skidding. Areal disturbance reported from selective logging in tropical rainforest range from 8 % (Marn and Jonkers (1982) to 30 % (Abdulhadi et al., 1981), much depending on extracted volume and harvesting technique. The clear-felling in catchment W5 including tractor extraction was, according to normal practice, a supervised uphill logging (Megahan and Schweithelm, 1983) carried out by a regular contractor and personnel involved in full scale operations in the SFI concession area. Major skid trail tractor tracks are shown in Figure 7. Of topographical reasons two stream crossings were made in W5 catchment. In this study of clear-felling, surfaces disturbed in W5 amounted to 24 % of the catchment area, while the corresponding figure for W4 with manual extraction was 4 % (I).

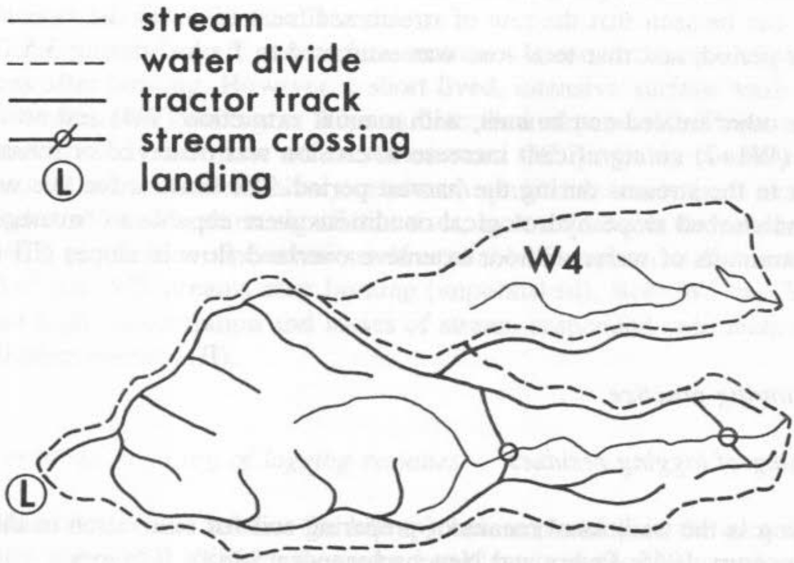

Figure 7. Layout of major tractor tracks in catchment W5 in Mendolong, Sabah, Malaysia.

As can be seen in paper I the mean effects on manual skid rails were low, while effects on tractor tracks were severe with strongly reduced bulk density and infiltrability. As pointed out earlier much of the disturbance along the tractor tracks consisted of removal of top soil combined with compaction. In this study the majority of disturbed surface was devoid of its organic layer and aggregated top soil. Extraction started in mid November 1987, all major skid trail tractor tracks in Figure 7 were completed in mid December 1987 and extraction continued until 18th January 1988 with a two week break for Christmas and New Year.

During the first month of extraction only a few minor rains (<20 mm) resulted in low impact of erosion on disturbed surfaces. At the first major rain of about 50 mm at the 2nd of December, however, extensive infiltration excess surface runoff occurred and the stream carried large amounts of sediments from surface erosion on tractor tracks. The right front cover photo on this thesis was taken on the 14th of December at "track 2" (Table III in VI) in the lowest part of W5 on sandy loam top soil. At that event gully erosion was noticed to start on several places along tractor tracks, during the second rain event of two of about 40 mm each in two days. Mean suspended sediment load in the stream was significantly increased for stormflows compared to baseflow concentration and compared to streams W4 and W6 (II). The difference in W5 baseflow concentrations were visible in the yellowish brown color of the water during this period, but not statistically significant as to concentrations. From Table 2 in paper II it can be seen that the rate of stream sediment loss was the largest during this period, and that total loss was estimated to 1 t ha^{-1} during 3.5 months.

In the other treated catchments, with manual extraction (W4) and no extraction (W1+2) no significant increase in erosion was observed or measured on soils or in the streams during the harvest period. The reasons for this was that the undisturbed slope hydrological conditions were capable to "manage" increased amounts of water without extensive overland flow in slopes (III and VI).

5.2 Preplanting practice

5.2.1 burning of logging residues

Burning is the traditional means of preparing soil for cultivation in the tropics (Boserup, 1965; Peters and Neuenschwander, 1988). It favors accessibility for cultivation, increases nutrient availability in the ash and the soil as well as it kills competing vegetation. This is also a concept adapted in forestry in many climates and silvicultural situations (eg. Goldammer, 1990). Paper V described why the burning of slash after clear-felling of tropical rainforest has many disadvantages compared to other situations, where fire fulfills silvicultural goals well without major adverse effects (eg. de Ronde et al., 1990). Due to the amount of fuel in the situation of slash left after the clear-felling of the rainforest, the fire may be harder and hotter compared to other environments. A harder burn is also desired to clear the soil. The consequence is a larger gaseous loss of Nitrogen and a whiter ash, devoid of Carbon and concentrated in other elements and nutrients. Gaseous losses were not quantified in this study, but has been reported to be proportional to weight losses of fuels (Raison et al., 1985). A harder burn also results in higher depletion of remaining slash and thereby lower amounts of slow nutrient release in the following years (Jordan, 1991). The warm humid environment is likely to be effective as to release nutrients from the ash by dissolving of salts, exchange processes and surviving or colonizing biological activity. Together with no living plants to take up nutrients and use water paired with short transit times of water (VI) leaching will be optimized. This experiment clearly showed large extra leaching losses after burning compared to no burning. Effect on both stream baseflow and stormflow concentrations was fast after burning (V). This was the situation, during the whole study, were the highest concentrations and highest rates of streamwater nutrient losses were recorded. The largest dissolved losses after burning were recorded from W1+2, even though the slash of that catchment had less biomass. That was concluded to be a consequence of a harder burn in

W1+2 paired with less disturbance to natural waterways and larger increase in stream stormflows in W1+2 (III and V).

Slope surface runoff and surface erosion was moderate on undisturbed soil surfaces after burning. However, a short lived, intensive surface wash of ash at the first rains after burning in W5 was described in paper VI. These small extra losses of eroded particulate ash to streamwater (VI) probably contained large amounts of nutrients. Preliminary results on qualitative analysis of stormflow suspended sediments show significant rises, to levels above the concentration in humus, in particulate concentrations of available Phosphorous (Sibbesen, 1983) in W1+2 and W5 streams after burning (unpublished). Both W5 and W1+2 showed high concentration and losses of stream suspended sediments in the period after burning (II).

5.2.2 removal or piling of logging residues

One of the main reasons for burning before planting in the area is that it is virtually impossible to reach the ground to plant in the vast amounts of slash left after clear-felling. Therefore, to avoid burning, either the logging residues has to be rearranged in piles or rows to get accessibility to plant or a larger amount of biomass has to be taken away in the harvest (Sim and Nykvist, 1991). In this experiment slash was rearranged into rows in W4, leaving strips of bare ground for planting, and burning was avoided (cf. 3.2 above).

A small increase in slope surface erosion in W4 may have been connected to an increased slope wash in the planting rows. However this 3 times increase in surface erosion must be considered moderate and no signs of rill erosion were observed in the planting rows (VI). No significant increases in stream suspended sediment concentrations were recorded during the time for planting and in the months following (II).

Rosén and Lundmark-Thelin (1987) presented investigations on increases in Nitrogen leaching from slash piles after clear-felling of boreal forest in Sweden (boreal humid climate (Strahler, 1978)). In that environment they found increased N concentrations in soil water under the slash, but decreasing concentrations in throughfall through the piles compared with rain water. They concluded the increased leaching to be an effect of enhanced mineralization in the humus layer under the piles and due to the fact that field vegetation could not establish in the soil covered by slash (cf. Bormann and Likens, 1970). Even though processes in the tropics are faster, that situation could to some extent be transferred to this experiment. In the boreal situation the losses were induced by the piling, but after clear-felling, in this situation the circumstances "in the pile" was apparent for the whole surface. Mineralization and leaching of the slash certainly had a quick start already in the slash itself above the ground, but either nutrients released there dripped down to the soil surface or sooner or later parts rich in nutrients (leaves and twigs) dried during sunny days and fell to the ground. Together with this supply of energy and nutrients, both moisture conditions (VI) and temperature were probably favorable for mineralization in the litter layer and the humus under the slash. As the slash after the clear-felling of the rainforest (and also the secondary vegetation in W1+2) completely covered the ground, conditions were not favorable for establishment of new vegetation. That was concluded to be the main reason for the high levels of leaching during harvest (cf above and V). That means that the rearranging of the slash into rows in W4 opened up soil surface for some secondary vegetation, which together with the planted trees could contribute to the immobilization of nutrients otherwise lost. To make the "piles" of slash even larger in between planting rows in this case did probably not make the situation much "worse" within and under the slash than it already was. Later, as the planted trees started to grow, the suppression of other vegetation by the slash was probably favorable for the trees (cf. 5.3.1 below).

An important difference between the non burning and burning situation was that the process of decomposition in and under the slash and the transfer of dead biomass (leaves, twigs and bark) to the biological mineralization at the soil surface was slower and continuous for a long time in the unburned slash. From San Carlos, Venezuela, Jordan (1989) pointed out that despite the fertilizing effect of burning it was rather the decomposition of the unburned slash that sustained soil fertility during three years of slash and burn cropping. Litter or mulch also acts as a source of energy for the below ground communities, enhancing nutrient conservation and plant growth (Jordan, 1991). In this study biomass relatively rich in nutrients like bark (Sim and Nykvist, 1991) and small branches were still observed to undergo continuous decomposition after several years, while most of this biomass was transferred to ash, or even lost in gaseous losses of Nitrogen in the burning. Without burning, also more of remaining vegetation already there in the slash and secondary vegetation in the planting rows had a better start to use nutrients released. These plants also reduced the increase in runoff (III) and probably thereby also larger leaching of nutrients (V).

Sim and Nykvist (1991) discussed the possibility to include smaller dimensions of stems and branches in the harvest, for example for energy purposes, to facilitate planting. That would induce a possible economic incitement for planting without burning, compared to the rather labor intensive method of clearing planting rows in the slash. However, as they pointed out, that would mean further removal of nutrients. Also such removal, if large, could reduce the favorable conditions described above for the planted trees. Such a treatment have to weigh the increased removal in harvest against the increase in loss from burning.

5.3 Period of plantation establishment

5.3.1 growth of planted forest and secondary vegetation

The growth of vegetation were expected to lead to an increased use of water and released nutrients, reflected in the increase in biomass of the plantations (cf. Table 1, 3.2 above). The annual effect of increase in runoff for W1+2 and W4 was less than 100 mm already in the third hydrologic year after clear-felling (89/90, III). For W5 on the other hand, the recession were broken between the second and the third year, resulting in an increase again in the third year. That later increase was also accompanied with a change in discharge dynamics towards faster runoff response at stormflows. In paper III it was explained that a result of gully erosion along tractor tracks was increased length of the ephermal channels of the stream, which was further described in paper VI.

As showed by Cunningham (1963) the rate of nutrient release (and consequently losses) will lessen with time. The greatest effects on nutrient concentration in streamwater from the treated catchments during harvest and preplanting described above, were rather short lived, and recession started when vegetation started to cover burned or disturbed surfaces (cf. Figure 1 and 2 in V). At this time also large amounts of ash were displaced by erosion and percolation (VI). This shortness of the bulk effect is also in accordance with earlier findings (V). Further in accordance with earlier studies of nutrient loss after logging (Zulkifli Yusop, 1989), concentration of some elements also related to geochemical sources like Potassium, Calcium and Magnesium had elevated concentration for a long time, in this study still after 3 years. In paper V a hypothesis is presented, that these prolonged losses of smaller amounts could be attributed to weathering in deeper soil horizons still out of reach for the roots of the newly established trees. That would mean that such a loss could be continuous in the case of a shallow rooted plantation crop. Zulkifli Yuosop (1989) reported similar raised concentration levels still 5 years after selective logging with soil disturbance in Berembun, Peninsular Malaysia, with a notation of the possibility of improved conditions for weathering after treatments as to soil temperature and moisture. Another complimentary reason for that observation might also be lower amounts of roots in the disturbed soil under the tractor tracks. In the view of the low conductivity of the sub soils in this study, another reason for prolonged high baseflow concentrations might be a delayed treatment response by deep soil water with much longer transit times. That is a process that well may be apparent together with the first hypothesis. However the fact that the prolonged raised concentration levels of Potassium, Calcium and other elements are accompanied by a much weaker trend of "protracted recession" by Nitrate, only originating from organic material, speaks for a higher importance for the first hypothesis.

For some of the elements of geochemical origin (K, Ca, Mg, Na, Si and S) paper 5 reported on negative post treatment net losses from W1+2, meaning smaller losses than if no treatment would have been applied as indicated by regression analysis with the control catchment W3 (cf Figure 6 in V and Table 3 in 6. below). This effect might have been due to an exhaustion effect of soil nutrients upon the repeated burning (V). Jordan (1991, after Buschbacher et al., 1984) also discussed how repeated burning of former forest land under pasture in the eastern Amazon region of Brazil results in smaller and smaller pulses of available nutrients, due to depletion of soils.

The increased concentrations of stream stormflow suspended sediment concentrations from W1+2 during and after burning, mainly consisting of ash, cease as vegetation recovered in late 1988 (II).

5.3.2 weeding

Type and number of weeding treatments were described above (3.2). No effects of weeding could be traced in surface runoff, surface erosion, stream suspended sediment or streamwater dissolved concentration. As the first circle weeding was made already in June 1988 (W1+2) - September 1988 (W5), that dead vegetation could have been expected to make some contribution to the streamwater nutrient loss, as other vegetation and planted trees at that time were little developed. If this process occurred, it was shaded by the high concentrations in recession since clear-felling and burning. At later weedings other vegetation probably had sufficient capacity to retain nutrients mineralized from dead weeds. Lundgren (1978) also discussed a possible positive effect of slash from weeding, as it could act as a mulch to lessen long term losses of soil organic matter and decrease maximum top soil temperatures (cf. Lal, 1974).

5.3.3 rehabilitation of disturbed soil

Tractor tracks were the only place where gully erosion could be observed. Data presented (VI) also show how the gully erosion still was active in late 1990. Apart from the considerably lowered production from the tractor extracted and burned catchment (Sim and Nykvist, 1991 and Table 1 in 3.2 above), the effect on runoff and erosion along the tractor tracks seem to be long lived as to infiltrability, runoff and gully erosion (I and VI). In many discussions the vegetation is said to cover the tracks quite quickly, indicating fast recovery from great erosion hazard. That is also the case after clearing without soil disturbance (eg. Northcliff et al., 1990). However, in this study, coverage was after 4 years in late 1991 still very incomplete in many parts. Also, what looked green and lush was still to a great extent vegetation expanding from the side, and almost nothing was rooted in the very tractor track soil, except for some vines and ferns. In late 1991, 4 years after start of treatment (almost half of the calculated rotation time for the forest plantation) the build up of a litter layer and the development of a change in soil structure due to the presence of organic material and activity had not started on most of the tractor tracks (cf. front cover photos). Only tracks with low disturbance (no excavation and only one pass by the tractor) showed some change in that aspect.

Stream stormflow concentration of suspended sediments showed a recession in the years after harvest and burning (II). In the third year after treatments, 1989/90, W5 stream stormflow suspended sediment concentrations were back to pre treatment levels (unpublished). That indicate that even though some gully erosion were still active, the majority of disturbed surfaces were stable to erosion despite the slow recovery in top soil properties, and only displacement of sediments within the catchments occurred after 1989. In 1988/89 W4 showed a significantly higher stream stormflow suspended sediment concentration (II), which was next year back to normal (unpublished). This temporary elevation of concentrations in the stream draining the "minimum disturbance" catchment may have been due to the increase in runoff leading to reactivation of sediments stored in the stream channel or by scour and channel widening at logging residues fallen into the stream (VI).

6. SUMMARY ON EFFECT ON DISSOLVED NUTRIENT LOSSES FOR THE FIRST 2.5 YEARS

6.1 Clear-felling and conversion to forest plantation

Table 3 present streamwater dissolved net losses as effect of treatments during different parts of treatments and the sum for 2.5 years compared with the amount of nutrients removed in the harvest. Note that the tabulated streamflow dissolved loss is the net loss induced by treatments as calculated by regression analysis compared to the discharge from control catchments (V).

Rates of loss effects were comparably lower in the first period of clear-felling and extraction. This can be seen as an effect of increasing amount of disturbance until culmination when the clear-felling and extraction were completed. Also number of rains were rather low during the first month of harvest, only 4 rains compared to a mean of 15 rains per month during 5 years (after VI). It is notable that soil disturbance in this study did increase the leaching both in stream water concentrations (Figure 6) and in effect of lost dissolved nutrients.

The highest rates, and in most cases also total effect, of losses occurred in the period right after burning and until establishment of a new vegetation cover. That was also the case for the unburned catchment W4, but rates and total losses were even larger from the burned catchments. To sum up earlier discussion, effects were as most intense during this period by combination of 1) largest amounts of nutrient rich parts of biomass under mineralization, 2) a minimum of vegetation to use released nutrients and 3) largest rate of runoff increase. This goes for all treatments, but burning enhanced all three of these components in the leaching process.

Total amounts lost were considerable compared with the content in easily decomposed parts of the biomass (V, after Sim and Nykvist, 1991). That comparison does not imply that it was exactly those elements that were lost. In consequence with above both increased mineralization of organic substances in the soil (5.2.2 above) and increased losses from weathering (5.3.1 above) contributed. Sim and Nykvist (1991) presented nutrient contents of the boles and bark of the W4 and W5 rainforest and Nykvist (1992) made comparisons of those data with other studies and discussed implications for nutrient removal in the harvest. Extracted wood volumes in percent of projected volume were 60.9 % (W4) and 50.5 % (W5) (Sim and Junim, 1990). The harvest nutrient removal in Table 3 is calculated as content in boles with bark times extracted percent. As some bark were scraped off during extraction these figures might be high.

Table 3. Dissolved nutrient losses during different conversions of tropical rainforest to forest plantation as effect of treatments (kg ha^{-1}) (after V), in Mendolong research area, Sabah, Malaysia. W1+2 was secondary vegetation, no extraction, burning and planting, W4 was lightly selectively logged forest, manual extraction and no burning before planting, W5 was lightly selectively logged forest, tractor extraction and burning before planting. (Some differences between tabulated sum and the sum of sub periods are due to round numbers in subperiods

form of loss and time period	element	net loss (kg ha^{-1}) W1+2	W4	W5
streamwater during harvest	N-tot	0.3	0.9	2.3
	P-tot	0.1	0.1	0.3
	K	3	10	18
4 months	Ca	-2[1]	2	4
	Mg	-2[1]	1	3
streamwater during and after burning	N-tot	13.0	7.2	10.2
	P-tot	1.1	0.2	0.6
	K	131	33	84
5 months	Ca	21	8	8
	Mg	1	3	5
streamwater during plantation establishment	N-tot	3.9	18.8	26.8
	P-tot	0.7	0.5	0.4
	K	-50[1]	44	64
24 months	Ca	9	14	15
	Mg	-4[1]	3	6
streamwater total 33 months	N-tot	17.7	27.0	39.9
	P-tot	1.8	0.8	1.3
	K	84	114	189
	Ca	28	25	27
	Mg	-5[1]	8	16
removal in the harvest	N-tot	-	142	118
	P-tot	-	3.0	2.5
	K	-	123	102
	Ca	-	242	201
	Mg	-	46	38

1. Negative number due to smaller loss than would have been the case without treatment, as indicated by regression analysis with control catchment.

By the separation in time of soil disturbance and burning it was possible to describe the effect of both soil disturbance and burning, even though these treatments were not separated in different catchments as to comparison of effects on forest conversion. However the separation of effects of each of these treatments from the effect of their interaction in W5 is more hypothetical. Total effect in dissolved losses were reduced to between 50 - 68 % for major plant nutrients by "minimum soil disturbance" and avoiding burning (W4), except for Calcium which experienced similar losses. This reduction of loss in W4 were of about the same relative size both during harvest and during and after burning compared to W5. It has been concluded that both soil disturbance and burning create large leaching losses, resulting in the largest total losses from W5 (V and above). On the other hand the sole burning without soil disturbance created the largest losses during and after burning from the secondary vegetation catchment (W1+2), despite that the amounts of biomass in W1+2 were so very much lower (3.2 above, after Sim and Nykvist, 1991). Even though the burning in W1+2 were more intensive (V) converting larger relative amounts of biomass to ash, the large loss at burning might be a sign that leaching after burning gives even larger losses when it is not combined with soil disturbance. Paper III indicated the non disturbed water ways in W1+2 to result in the largest increases in stream stormflows. Also in paper VI it has been indicated that the shallow throughflow in undisturbed soils, generating stream stormflows and high baseflows, were contributing to the high increases in stream water concentrations during treatments, while paper III described how the soil disturbance in W5 contributed to concentrate the increase in runoff to stream baseflows and consequently longer transit times. Finally, paper VI underline the importance of the quality aspect of lost sediments. When the content of available nutrients lost in stream suspended load (II) is also considered, the relative nutrient loss from burning will probably be even larger, as indicated by preliminary results from qualitative sediment analysis (end of 5.2.1 above).

Consequently, this discussion indicate that minimum soil disturbance combined with subsequent burning might create larger leaching losses than might soil disturbance without subsequent burning before planting, under similar time and environmental circumstances of this study. However, this is a hypothesis compared to the absolute result above of the large reduction of losses from the combination of minimum soil disturbance and no subsequent burning, which stand out as the major finding in the part of the thesis in this study. Further studies should concentrate on process studies to verify the hypothesizes above to further increase transferability of results from this catchment study.

The results on streamwater losses have implications on the first and later harvests of the plantation. The residual biomass after the harvest of the plantation can be expected to be smaller than at the conversion of the rainforest.

Also soil disturbance by tractors (if used for extraction) can be lowered by using already existing tracks not rehabilitated after the conversion. These circumstances indicate that leaching losses probably will be smaller at the first harvest compared to the conversion of the rainforest. However the principles and effects of reducing adverse impacts implied by this study by minimizing soil disturbance and avoiding burning will be valid also for the harvest and regeneration of the plantation. The smaller dimensions and biomass volumes of the plantation will also more easily allow the use of cable systems for extraction and planting without preceding burning of slash.

This study and this thesis mainly deal with on site effects for the sustainability of forest production. However, many off site effects as siltation of rivers, pollution and eutrophication of receiving aquatic ecosystems are to a high degree functions of streamwater characteristics studied here. It is essential to point out that the reduction of on site effects also result in the similar reduction in off site effects. Above it was pointed out that for example sediments from stream bank erosion might not be such a large loss of nutrients from the site as by the loss of ashes. However in the off site use of water for fishery or raw water for industrial purposes, the mineral particles in suspension also very much contribute to the problems of siltation.

It can be argued that areas of natural high sediment yields will suffer little if any extra effect from human influence, compared with other sites with low natural denudation rates (Bruijnzeel, 1990). However, rare events can be devastating in landscapes with high intensity land use and dense population (eg. Trustrum et al., 1990). In determinating the effect of high intensity disturbances by man, the extra effect of the 50- or 100-year return high magnitude event in combination with human interference will be very difficult to access. Rather, the research on quantification of effects of land use is directed to deal with "everyday" moderate an high frequency effects. By recognizing "acceptable" levels of erosion and siltation of rivers for sustainable resource use and implementing them in land use planning, will of cause also contribute to reduce extra effects in the case of the rare high magnitude event.

6.2 Comparisons with selective logging

The objective of this thesis is to describe some effects of clear-felling and conversion to forest plantation (cf. 2.3 above). However, as selective logging is a much more common forestry practice, and studies of effects comparable with this study are seriously lacking, except for a few exceptions (eg. Zulkifli Yusop, 1989 and 1991; Douglas et al., 1992), some brief comparisons can be worth wile where such can be considered possible (cf. VI).

The first two months of the harvest in W5 were rather similar to a selective logging operation with high volume of extraction, as the whole catchment were affected at the same time. The clear-felling was not operated as a "front of felling". That means that the effects during the harvest can be used as a rough estimate of the effects of a selective logging operation in the region on similar soils. In consequence with the findings in this study the total effects of a selective logging can be assumed to be less than from a clear-felling operation due to 1) less biomass subject to decomposition, 2) much more living vegetation and undisturbed root systems to use released nutrients and 3) smaller increase in runoff. However, like in this study, dissolved nutrient losses, like other on site effects reducing regeneration and future production, can probably be considerably lowered by reducing amount of soil disturbance.

As pointed out in paper VI, in the effects of soil disturbance on hydrology and erosion, on process level, there are also possibilities of comparisons between selective logging and the data from the soil disturbance in this study.

7. CONCLUSIONS

Treatment induced streamwater net losses of dissolved nutrients from this study were high. The rate of these large streamwater losses were largest after burning of logging residues, and total streamwater losses were larger from treatments including burning. The largest loss occurred from the combination of soil disturbance and subsequent burning (normal practice). Under forest cover shallow throughflow with short transit times of water contributed to nutrient conservation. After clear-felling on the other hand, the maintenance of this mechanism in undisturbed soils contributed to the large losses of dissolved nutrients from decomposing slash and dead top soil roots, and the concentration of losses to stream stormflows. Treatments did not lead to extensive slope surface runoff and surface erosion on undisturbed soil surfaces. However a small increase in surface erosion, carrying ashes after the burning, probably carried large extra amounts of nutrients in particles.

Disturbed soils, on the other hand, had extensive surface runoff during storms and resulted in the largest and most prolonged increases in runoff. Soil disturbance was also related to increased nutrient leaching by losses of nutrients from disturbed organic top soil mixed with other soil and logging residues, together with the worst circumstances for nutrient retention. Gully erosion on tractor tracks were the main source for heavy siltation of the stream.

The serious effects on soil and water were strongly reduced with the "minimum disturbance" treatment avoiding soil disturbance by tractors and burning of residues. Increases in runoff, stream siltation and streamwater dissolved losses were reduced about 50 % with manual extraction and no burning compared to normal practice of tractor extraction and burning before planting. Even though the manual extraction used in the "minimum disturbance" treatment may be slow and laborious, these findings above clearly shows the importance to adjust and develop silvicultural methods towards a reduction of adverse impacts on site and off site and there by increasing future yields from plantations, and to reduce undesired off site effects.

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Soil disturbance and loss of infiltrab[illegible] by mechanized and manual extraction of tropical rainforest in Sabah, Malaysia

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ABSTRACT

Malmer, A. and Grip, H., 1990. Soil disturbance and loss of infiltrability caused by mechanized and manual extraction of tropical rainforest in Sabah, Malaysia. For. Ecol. Manage., 38: 1–12.

The rate of infiltration, or infiltrability, is known to change due to soil disturbance. The infiltrability is normally higher in the initial stages of infiltration before wetting of the soil, while pressure gradients predominate over the gravitational forces. This is described by the term 'sorptivity' which is the cumulative infiltration as a function of the square root of time, which often is constant during early stages of infiltration. As gravitation starts to dominate, infiltrability decreases to a constant, steady-state infiltrability. Drastic changes in infiltrability may lead to overland flow causing erosion and nutrient leaching.

Dry bulk density, steady-state infiltrability and sorptivity were measured in reference forest and after manual and crawler-tractor timber extraction in different watersheds on Orthic Acrisol (40% clay, 40% sand) and Gleyic Podsol (10% clay, 80% sand). Six years old tractor tracks after selective logging were also investigated.

New tractor tracks covered 24% of the mechanically extracted area, while skid trails covered 4% of the manually extracted area. In the top 10 cm, mean dry bulk density increased after all treatments, but was significant only in the top 5 cm on clay soil after tractor extraction, where it increased from 0.82 to 1.23 g cm^{-3}.

Means of steady-state infiltrability and sorptivity were lower on treated areas than in the forest. Differences of mean steady-state infiltrability were significant for all tractor treatments compared with reference forest on both soil types and also with manual extraction on clay soil. On clay soil (and sand) mean steady state infiltrabilities were 154 (48.7), 36.7 (11.6), 0.28 (1.26) and 0.63 (–) mm h^{-1} in forest, on manual and on new and old tractor tracks respectively.

INTRODUCTION

Use of heavy machinery in mechanical clearing and timber extraction in forests is known to cause removal, disturbance and compaction of soil (Greacen and Sands, 1980; Soane, 1986; Incerti et al., 1987; Kamaruzaman, 1988; Ole-Meiludie and Njau, 1989). Several studies have reported change in phys-

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New tractor tracks covered 24% of the mechanically extracted area, while skid rails covered 4% of the manually extracted area. In the top 10 cm, mean dry bulk density increased after all treatments, but was significant only in the top 5 cm on clay soil after tractor extraction, where it increased from 0.82 to 1.28 g cm^{-3}.

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ical properties of tropical soils after forest clearing for agricultural purposes (Van der Weert, 1974; Seubert et al., 1977; Lal and Cummings, 1979; Hulugalle et al., 1984; Lal, 1984; Dias and Nortcliff, 1985; Alegre et al., 1986). One serious consequence of soil compaction is reduction in soil permeability. This causes accelerated runoff which leads to erosion (Lal, 1986) and an increase in nutrient leaching (Kang and Lal, 1981). The potential loss of favorable soil structure and valuable nutrients can be seriously detrimental to the prospects for future sustained production on the areas involved (Couper et al., 1981).

This paper present results on change in steady-state infiltrability, sorptivity, soil dry bulk density and areal percent of disturbed soil in two clear-felled watersheds where different extraction methods were used.

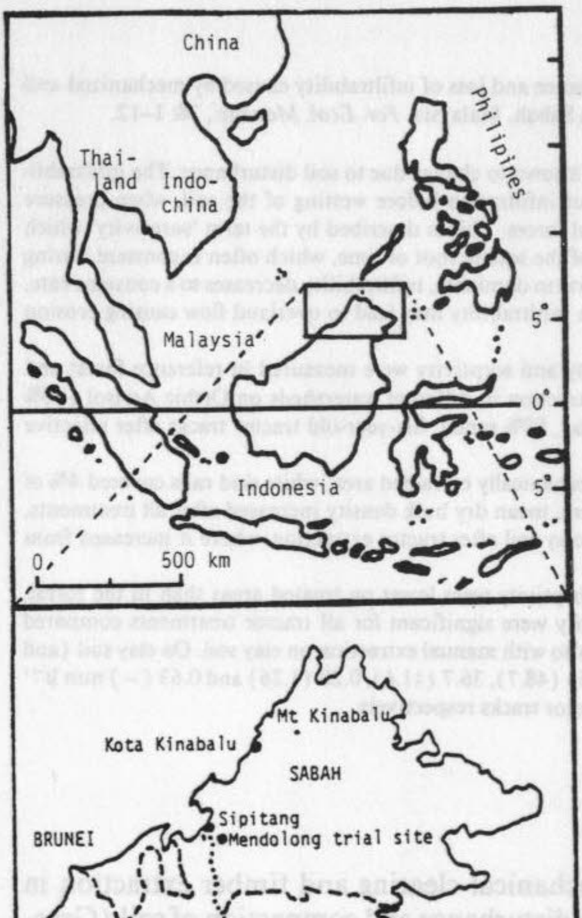

Fig. 1. Location of Mendolong trial site.

RESEARCH AREA

The study is part of a paired watershed-impact study on the consequences of clear-felling and plantation forestry at Mendolong in the Sipitang area, Sabah, Malaysia. The five watersheds used in the impact study are situated in logged-over dipterocarp forest (selectively logged in 1981) at 650–750 m above sea level on the foothills of Mount Lumako, 35 km south-east of the coast at Sipitang (115.5 E, 5.0 N, Fig. 1). Slopes in the area are mostly moderate, ranging up to 27%, but steeper slopes of up to 57% also occur. Effective soil depths vary between 1 and 3 m. The present study was carried out within two of the watersheds of the impact study, W4 (3.4 ha) and W5 (9.7 ha).

Two different soil types can be recognized in the area (Wong, personal communication, 1987), namely Orthic Acrisol with high clay content and a Gleyic Podsol with lower clay content and high sand content (Fig. 2). Porosity in the uppermost 20 cm ranges from 59 to 67% in the uppermost 20 cm of the Acrisol, and from 45 to 54% in the Podsol respectively (Malmer, unpublished data, 1988). The areal coverage was estimated to 66% Acrisol and 34% Podsol at W4 and 38% Acrisol and 62% Podsol at W5. The humus layer was between 1 and 3 cm deep on both soil types.

Areas at high elevation at the west side of the Crocker Range have the highest annual precipitation in the region (Hing, 1986; Walsh, 1982). Mean yearly precipitation at the trial site was 3850 mm (August 1985–July 1988) and mean yearly runoff was 1580 mm before clear-felling (August 1985–July 1986) (Malmer, unpublished data, 1988).

The watersheds were selectively logged in the middle of 1981. At that time crawler tractors were used for extraction. In 1987, tracks from this treatment

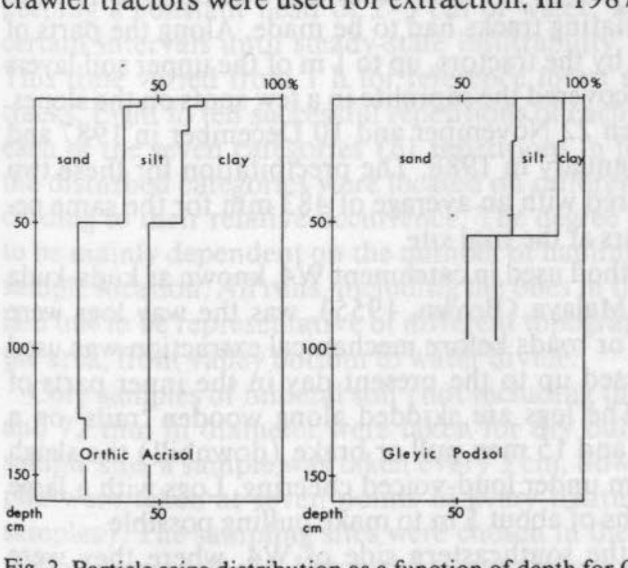

Fig. 2. Particle seize distribution as a function of depth for Orthic Acrisol and Gleyic Podsol at Mendolong, Sabah, Malaysia (after Wong, personal communication, 1987).

were still recognizable in some parts of the watersheds. The clear-felling for the present study was conducted during November 1987 and January 1988.

MATERIALS AND METHODS

Clearing and extraction

The different extraction methods used in this study were meant to show effects of the normally used method in the area, and also of a method with less soil disturbance. In both watersheds felling was done manually using chain saws, first by cutting timber and pulpwood and later, after extraction, by slashing the remaining vegetation.

The normal practice for extraction in the area is hauling with crawler tractors, and this method was used in watershed W5. The tractors used were light bulldozers (D4 and D6 class) with straight blades. Vegetation and topsoil were pushed to the sides along the tracks to ease the hauling of the logs. The logs were hauled without sledge or skidding cone. Due to slopes and the size of the full tree-length logs, only one log was normally hauled at each trip by the tractor. Uphill logging (Megahan and Schweithelm, 1983) was used and the landing site was located mainly outside the water-divide by the road side at the highest point of W5. The main access track followed the north-west water-divide of W5. During rain the extraction halted, and heavy storms normally stopped work for one or two days due to the tracks being too slippery. Occasionally, after longer periods of heavy rain, some parts of the tracks were filled with water or were too slippery to pass. In such a case more soil had to be pushed aside, or new deviating tracks had to be made. Along the parts of the tracks most heavily used by the tractors, up to 1 m of the upper soil layers were pushed aside, which uncovered the saprolite in a few spots on the slopes. All hauling was done between 22 November and 10 December in 1987 and between 5 January and 18 January in 1988. The precipitation for these two months was 533 mm compared with an average of 483 mm for the same period during the last three years at the trial site.

The manual-extraction method used in catchment W4, known as kuda-kuda in Sabah and 'panglong' in Malaya (Brown, 1955), was the way logs were normally extracted to rivers or roads before mechanical extraction was used in Borneo. The method is used up to the present day in the inner parts of Sarawak and Kalimantan. The logs are skidded along wooden 'rails' on a wooden sleigh. Between ten and 15 men pull or brake (downhill) the sleigh with ropes in a steady rhythm under loud-voiced cheering. Logs with a large girth had to be cut into lengths of about 5 m to make pulling possible.

All logs were skidded to the southeastern side of W4, where they were winched up by crawler tractor and hauled to the landing site by the road side.

The tractor hauling of the W4 logs was done with the same tractors that were used for the W5 logs. (In this case several logs were hauled at the same time due to the short length of the logs.) The tractors used tracks at the water-divide between W4 and W5 and the main track along the northwest water-divide of W5. Tractors only occasionally happened to move marginally inside the W4 watershed.

Area affected

The total length and width of skid rails and tractor tracks inside W4 and W5, respectively, were recorded and mapped. From this the total area and relative area affected in the two catchments was calculated. At this time notes on the relative degree of disturbance along the tracks were also made.

Observations of soil physical characteristics

Field measurements of infiltration and dry bulk density were mostly made between January and March 1988, on seven different combinations of soil type and treatment. The seven categories tested were: (1) reference forest on clay soil; (2) reference forest on sandy soil; (3) tractor tracks in W5 on clay soil; (4) tractor tracks in W5 on sandy soil; (5) skid rails in W4 on clay soil; (6) skid rails in W4 on sandy soil; and (7) old (1981) tractor tracks on clay soil.

Cumulative infiltration as a function of time was measured in the field using a double-ring infiltrometer with 156 mm and 256 mm inner diameters, keeping a constant head of 2–3 cm of water. Measurements were taken at certain intervals until steady-state infiltrability (Hillel, 1980) was reached. This time varied from 1 h for reference forest soil to 4 h on soil in tractor tracks. Eight to ten successful repetitions of each measurement were done on each of the seven categories (61 repetitions in total). The measurements in the disturbed categories were located on different degrees of disturbance according to their relative occurrence. The degree of disturbance was assumed to be mainly dependent on the number of haulings or skiddings made at each sample location. All runs, including the ones in the reference forest, were also laid out to be representative of different topographical positions occurring in the area, from valley bottom to water divide.

Core samples of mineral soil (not including the humus layer) 50 mm deep and 72 mm in diameter were taken for dry bulk density analysis. At every sample site, a sample was taken every 5 cm, down to a depth of 20 cm. Samples were taken at seven points in every treatment category (totalling 196 samples). The sampling sites were chosen in the same way as those for infiltration measurement. Dry-weight analysis was made at the Mendolong soil laboratory.

Analysis

The rate of infiltration, or infiltrability, was calculated from the cumulative infiltration as a function of time. The infiltrability is high during the early stages of ponded infiltration when the pressure gradients predominate over the gravitational forces, but decreases gradually to a constant rate, the steady-state infiltrability (Hillel, 1980), when gravitation is the only acting force. Philip (1957a) used the term sorptivity for the cumulative infiltration as a function of the square root of time during the initial stages of infiltration. The sorptivity is often constant during this time-period.

Sorptivity and steady-state infiltrability were determined according to Collis-George (1980). All data for physical properties were tested for normality and variance. Differences between means at different treatment categories were tested using a nonparametric multiple test (Dunn, 1964).

RESULTS

Area affected

The areal coverage of new tractor tracks in the mechanically extracted watershed was 24%. Coverage of manual skid rails in the other watershed was 4%. The areal coverage of old tractor tracks after the 1981 selective logging was not surveyed, but was considered small in comparison with the area of new tracks.

Steady-state infiltrability and sorptivity

In this study, data from different treatments were not normally distributed. Log-transformation of the data did not result in a normal distribution. The variances between the groups were also unequal. Therefore a nonparametric test was used to differentiate between treatment categories.

On clay soil the mean value of steady-state infiltrability was 154 mm h^{-1} in the forest but only 0.28 mm h^{-1} on new tractor tracks (Fig. 3). On the six-year-old tractor tracks the mean steady-state infiltrability had recovered a little and was 1.26 mm h^{-1}. The mean steady-state infiltrability on sandy soil was 48.7 mm h^{-1} in the forest and 1.26 mm h^{-1} on new tractor tracks.

The sorptivity mean followed the same pattern on clay soil, with high values in reference forest (104 mm h^{-1}) and low values on new tractor tracks (1.08 mm h^{-1}; Fig. 4).

Mean steady-state infiltrability and mean sorptivity were significantly lower on old and new tractor tracks compared with reference forest and manual skid rails on clay soil. On sandy soil, the only significant difference was be-

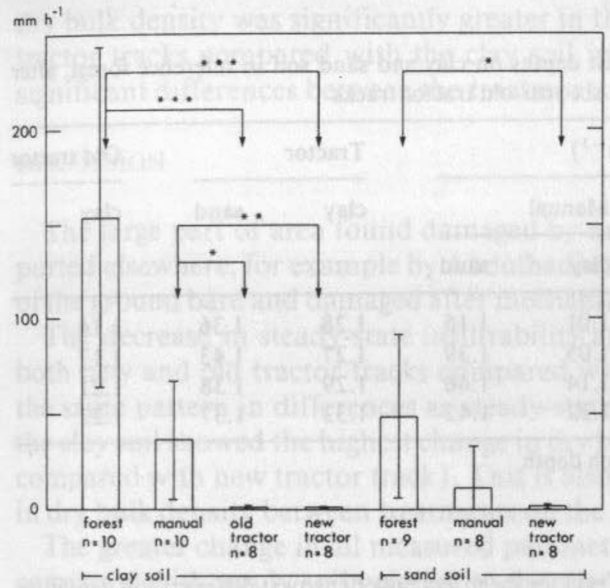

Fig. 3. Mean steady-state infiltrability on clay and sand soil (with standard deviation bars) in reference forest, after manual and tractor extraction and on six-year-old tractor tracks at Mendolong, Sabah, Malaysia, *'s, significance in differences between treatment categories on the respective soil type according to Dunn multiple test. $n=8$–10 for each treatment category.

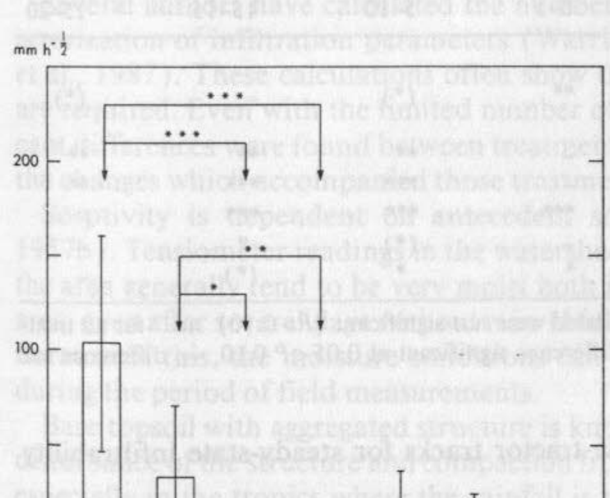

Fig. 4. Mean sorbtivity on clay and sand soil (with standard deviation bars) in reference forest, after manual and tractor extraction and on six-year-old tractor tracks at Mendolong, Sabah, Malaysia. *'s, significance difference between treatment categories on the respective soil type according to Dunn multiple test. $n=8$–10 for each treatment category.

TABLE 1

Means of dry bulk density at different depths on clay and sand soil in reference forest, after manual and tractor extraction and on six-year-old tractor tracks

Depth (cm)	Dry bulk density (g cm^{-3})						
	Forest		Manual		Tractor		Old tractor
	clay	sand	clay	sand	clay	sand	clay
0– 5	0.82	1.12	1.01	1.15	1.28	1.36	1.16
5–10	0.89	1.36	1.05	1.39	1.27	1.43	1.17
10–15	0.95	1.46	1.14	1.46	1.29	1.38	1.21
15–20	1.05	1.52	1.22	1.42	1.33	1.37	1.22

$n=7$ for all treatment categories at each depth.

TABLE 2

Statistical analysis[1] (Dunn Multiple Test) for mean dry bulk density at different soil depths on clay and sand soil in reference forest, after manual and tractor extraction and on six-year-old tractor tracks

	Soil depth (cm)			
	0–5	5–10	15–15	15–20
Clay soil				
Forest < tractor	**	(*)	–	(*)
Clay/sandy soil				
Forest (clay) < forest (sand)	–	**	**	**
Forest (clay) < manual (sand)	–	***	***	**
Forest (clay) < tractor (sand)	***	***	***	–
Manual (clay) < manual (sand)	–	(*)	*	–
Manual (clay) < tractor (sand)	*	*	(*)	–

[1]Combinations of differences not tabulated were not significant ($P>0.10$). $n=7$ for all treatment categories at each depth. (*), difference significant at $0.05<P$ 0.10. —, difference not significant ($P>0.10$).

tween reference forest and new tractor tracks for steady-state infiltrability (Figs. 3 and 4).

Dry bulk density

The greatest change in mean dry bulk density, compared with the undisturbed forest soil, occurred in the uppermost 5 cm on the treated soils (Table 1). In the undisturbed forest there was a significant difference in mean dry bulk densitv between the two soil types but not in the uppermost 5 cm. Mean dry bulk density was significantly greater in the uppermost 5 cm on the new tractor tracks compared with the clay soil in the forest, but there were no significant differences between the treatments on sandy soil (Table 2).

DISCUSSION

The large part of area found damaged by mechanized logging was also reported elsewhere, for example by Abdulhadi et al. (1981), who reported 30% of the ground bare and damaged after mechanized logging in East Kalimantan.

The decrease in steady-state infiltrability and sorptivity was dramatic on both new and old tractor tracks compared with the forest. Sorptivity shows the same pattern in differences as steady-state infiltrability. The top layer of the clay soil showed the highest change in dry bulk density (clay soil reference compared with new tractor track). This is also the only significant difference in dry bulk density between treatments on the same soil type.

The greater change in all measured parameters after treatment on clay soil compared with sandy soil was due to the aggregated structure and high root density of the topsoil of the clay soil, which was disturbed or scraped away by the tractors. Aina (1984) has shown the significance of the role played by earthworms in determining soil structure, and hence infiltration rate in forest soil in Nigeria.

Several authors have calculated the number of samples required for characterization of infiltration parameters (Warrick and Nielsen, 1980; Sharma et al., 1987). These calculations often show that a large number of samples are required. Even with the limited number of samples in this study, significant differences were found between treatments, indicating the magnitude of the changes which accompanied those treatments.

Sorptivity is dependent on antecedent soil moisture content (Philip, 1957b). Tensiometer readings in the watersheds showed that the clay soils in the area generally tend to be very moist both in the forest and in the cleared area, even after several days without rain (Malmer, unpublished data, 1988). Because of this, the moisture conditions can be regarded as fairly constant during the period of field measurements.

Bare topsoil with aggregated structure is known to lose infiltrability due to disturbance of the structure and compaction from rain drops (Hudson, 1984), especially in the tropics where the rainfall is more intense. In this study, no comprehensive comparisons of infiltrability were made on cleared, or cleared and burned, areas not affected by tractor tracks or skid rails. To get an approximate idea of this change, several test runs were made, at different times, on soil in the cleared and burned area on both clay soil and sandy soil. In March 1988 these tests did not indicate any difference relative to the reference forest, and in November 1988 the tests indicated even higher infiltrability in these areas. This would indicate that any increase in overland flow would be caused mainly by the disturbance made by the different extraction methods compared above. The absence of change of infiltrability from raindrop impact in these tests could be explained by rapid growth of vegetation covering the bare soil. Calcium in the ash could also be favorable for the aggregation of the top soil, improving infiltrability.

It should be noted that the changes in dry bulk density on treated soils, presented in this study, are not only caused by compaction. When the tractors push topsoil to the sides, deeper soil layers with naturally higher dry bulk density and lower infiltrability are uncovered to make up the new 'topsoil' on the tractor track. This study only considers the total effect from the disturbance caused by the extraction. The total loss of infiltrability for the watershed is also the main factor governing the surface runoff and thereby erosion and nutrient leaching.

CONCLUSION

Logging operations are often said to reduce the soils ability to infiltrate and distribute water and to reduce root penetrability, but few thorough studies have been reported from tropical regions. This study points to a considerable change in dry bulk density and infiltrability on clay soil where tractors are used for the extraction of wood. The tractors degraded one-quarter of the logged area. This would lead to a reduction in total infiltration capacity of about 25% for a watershed wholly on clay soil (considering only steady-state infiltrability).

The non-significant difference between infiltrability of the forest and the manual skid rails also suggests that it is the load of the heavy tractors and the disturbance and removal of the uppermost soil layers that causes the changes. Also notable is the very low infiltrability on old tractor tracks from the selective logging. After six years (more than half the calculated rotation period for a forest plantation in this region) there is practically no difference in infiltrability between these tracks and the new ones. These results show the importance of extraction systems that minimize the area affected and the damage to the topsoil. Cable yard logging (Miller and Sirois, 1986) or manual extraction systems would be desirable on these soils.

Loss of infiltrability and clearing of forest are known to cause increases in runoff which lead to increased erosion and nutrient losses (Lal, 1986; Kang and Lal, 1981). The maintenance of desirable site qualities in tropical areas is of ultimate importance not only for sustained on site production but also to limit the demand for the last virgin forests which remain.

ACKNOWLEDGEMENTS

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Research Needs and Applications to Reduce Erosion and Sedimentation in Tropical Steeplands (Proceedings of the Fiji Symposium, June 1990): IAHS-AISH Publ. No. 192, 1990.

Stream suspended sediment load after clear-felling and different forestry treatments in tropical rainforest, Sabah, Malaysia

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ABSTRACT A paired watershed experiment [illegible] in logged-over tropical rainforest in [illegible] Malaysia to study the effects of different [illegible] to plantation forest on suspended sediment [illegible]. Four watersheds were clear felled: W1 and W2 (secondary vegetation after forest fire) from which no logs were extracted; W4 where manual extraction was practised; and, W5 where normal technique of tractor extraction was used. After felling, the basins were planted with plantation forest. At W1, W2 and W5 this was preceded by burning of the slash. Two catchments were monitored for reference. Suspended sediment soil loss was determined to be 3.9, 2.1 and 0.4 t ha^{-1} following tractor extraction, manual extraction and no extraction respectively for the first 12.5 months during and after extraction.

INTRODUCTION

Erosion following logging, causing high loads of suspended sediments in streams and rivers, is a great problem in many parts of the world. The risk for erosion is especially high in tropical steeplands with large amount of precipitation and fine grained soils with low organic content (Sundborg, 1956) when the protective forest cover is removed and topsoil disturbed (Lal, 1986).

There are quite a number of studies on suspended load from undisturbed tropical forests (Van Dijk and Ehrencron, 1949; Bell, 1973; Douglas, 1973; Turvey 1975; Ree, 1981; Bailly, 1984) and also on the effect of deforestation and conversion to agricultural land (Kellman, 1969; Cooper et al., 1981). Long term studies on the effect of different forestry treatments in the tropics are still very scarce. A few studies on sediment load from tropical tree plantations are reported (Richardson, 1982; Bruijnzeel, 1983; Amphlett, 1984). Roche (1981) and Fritsch (1983) reported on studies in small basins in French Guyana where silvicultural treatments were included in the study of the effects of different management practices. Similar studies have been carried out in Queensland (Cassells et al., 1982). A study on suspended load after selective logging was also carried out in Sabah by Greer et al. (1989 and this volume).

This paper presents the estimation of suspended sediment load carried by streams from six paired watersheds in Sabah, Malaysia before, during and the first year after different forestry treatments.

The study is part of an extensive long term study on changes in

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ABSTRACT A paired watershed experiment was carried out in logged-over tropical rainforest in south-west Sabah, Malaysia to study the effects of different conversions to plantation forest on suspended sediment soil loss. Four watersheds were clear-felled: W1 and W2 (secondary vegetation after forest fire) from which no logs were extracted; W4 where manual extraction was practiced; and, W5 where normal technique of tractor extraction was used. After felling, the basins were planted with plantation forest. At W1, w2 and w5 this was preceded by burning of the slash. Two catchments were monitored for reference. Suspended sediment soil loss was determined to be 3.9, 2.1 and 0.9 t ha^{-1} following tractor extraction, manual extraction and no extraction respectively for the first 18.5 months during and after extraction.

INTRODUCTION

Erosion following logging, causing high loads of suspended sediments in streams and rivers, is a great problem in many parts of the world. The risk for erosion is especially high in tropical steeplands with large amount of precipitation and fine grained soils with low organic content (Sundborg, 1956) when the protective forest cover is removed and topsoil disturbed (Lal, 1986).

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This paper presents the estimation of suspended sediment load carried by streams from six paired watersheds in Sabah, Malaysia before, during and the first year after different forestry treatments.

The study is part of an extensive long term study on changes in hydrology and nutrient budgets when different silvicultural methods are used to establish tree plantations.

RESEARCH AREA

The six watersheds used for this impact study are situated at 650-750 m a.m.s.l. on the foothills of Mount Lumako 35 km south east of the coast at Sipitang (115.5°E, 5.0°N) in Sabah, Malaysia. The size of the watersheds are between 3.4 and 18.2 ha, and the treatments being compared cover 100% of the respective basins.

Slopes in the area are mostly moderate, ranging up to 27%, but steeper slopes of up to 57% also occur. Two different soil types can be recognized in the area (Wong, personal communication, 1987), namely Orthic Acrisol with high clay content and a Gleyic Podsol with lower clay content and high sand content (after Wong, Malmer & Grip, in press).

Porosity in the top soil and steady state infiltrability are higher in the Acrisol in undisturbed forest due to better developed soil structure. This is drastically changed after tractor disturbance when mean steady state infiltrability is lowered from 154 to 0.28 mm h^{-1} (Malmer and Grip, in press).

The precipitation at the trial site is mostly comprised of convective rain, but there is also an influence of monsoonal rain. This monsoonal rain in the area is more pronounced in a gradient to the coast (Wong, 1989).

MATERIALS AND METHODS

Treatments

The different pretreatment characteristics and the treatments of the catchments were:

(a) Three watersheds with selectively logged forest (1981) struck by forest fire 1982/83 (Beaman et al., 1985) The burned area was comprised of a mix of surviving trees and secondary vegetation. At W1 and W2 100% of the area was accidently burned and at W3 80%.

(i) W1: Non-mechanized clearing of remaining trees and secondary vegetation, burning of slash and planting.

(ii) W2: The same treatment as W1.

(iii) W3: Reference watershed. No treatments.

(b) Three watersheds with forest selectively logged in 1981.

(i) W4 Manual felling, manual extraction of logs, moving of slash into rows and planting in between the rows, without burning.

(ii) W5 Manual felling, extraction using crawler tractors, burning of the slash and planting (normal practice).

(iii) W6 Reference watershed. No treatments.

Watersheds W1 and W2 were treated as one because of uncertainty in identifying the phreatic water divide between the two streams.

Normal practice for extraction in this region is crawler tractor hauling of the logs. The coverage of tractor tracks were 24% of the area of W5 after the treatment (Malmer and Grip, in press). The manual extraction method used in catchment W4 is known as "kuda-kuda" in Sabah (Brown, 1955; Malmer & Grip, in press).

The normal practice of burning remaining slash was used in W1, W2 and W5. About one month after burning, grasses and weeds started to colonize the soil, and after three months most soil except for tractor tracks was covered with vegetation.

At W4 burning was avoided and all slash were cut into smaller constituents and put into rows, leaving strips of bare ground for planting. Planting was carried out at W1, W2 and W4 in mid March and W5 was planted in the beginning of May 1988.

Hydrological monitoring

Precipitation has been recorded weekly since April 1985 from 12 rain gauges throughout the research area. One hourly recording rain gauge was used also to determine the amount of rain during different rain events.

Runoff from the watersheds has been assessed by use of Ott water stage recorders in stilling wells connected to the stream upstream from 120° angle glassfibre reinforced plastic flumes installed on clay beds. The water stage recording has been continuous since mid 1985. The last runoff data used for this study was from May 1989. Periods of missing data were filled in using regression equations between water flow in different streams.

Suspended load sampling and analysis

One litre samples of stream water taken from the outflow jet from the flumes have been used for filtration. For the period of this study (August 1986 - August 1989) 2187 samples were taken in the six streams. Three to nine times for every watershed, detailed sampling every 15 or 20 minutes has been carried out during rainstorms. During the period of treatments one high flow sample per watershed and storm was taken once or a few times a week.

Samples were filtrated in the field using Munktell OOH filter papers. Later the filters were oven dried at 105°C over night and weighed to the nearest 0.1 mg after cooling in a dessicator. The filter papers were not pre-weighed. Instead dry weight was analyzed for 164 not used filter papers. The full variation within one box of filter papers and samples from 26 different boxes were included in this analysis. Mean paper weight was 789.1 mg, the standard deviation was 9.3 mg and the maximal deviation 26.6 mg.

Data analysis and load calculation

Distributions and statistics were estimated from data on mean values of suspended load concentrations. Concentration data was divided into "low flow samples" (lf) and "high flow samples" (hf) and into four different time periods, i.e.:

(a) 1: August-October 1986 and May-November 1987, totalling 10.5 months before treatments;

(b) 2: November 1987-February 1988, 3.5 months during the intensive treatment period;

(c) 3: March 1988-July 1988, 5 months from burning to the appearance of vegetation cover;

(d) 4: August 1988-May 1989, 10 months in the post treatment period.

The significance of differences between period mean lf-concentrations and hf-concentrations for the same or different watersheds were calculated using Tukey-Kramer multiple test (Kramer, 1956).

Linear regressions between stream flow and suspended sediment concentration have been calculated for non transformed and log-transformed data and also by separating rising and falling limbs of the hydrograph.

Graphs of sediment concentration and stream flow during rain events and hysteresis loops for the same parameters have been studied to determine characteristic responses of the streams. Total suspended sediment load for separate rain events has been calculated by using data from the same rain events.

Total load of suspended sediment carried per hectare by the streams during different periods has been calculated. For streams and periods with no significant difference between lf- and hf-concentrations, load was calculated by multiplication of mean lf- and hf-concentrations with respective lf- and hf-stream flow volumes for the four periods. Where there were significant differences between lf- and hf-concentrations and significant regressions between concentration and streamflow were found, these regressions were used to calculate the total load from the continuous stream flow records.

RESULTS

Hydrologic regime

The mean precipitation for the hydrologic year of the treatments (August 1987-July 1988) for all 12 rain gauges at the trial site was the highest during the four years of measurements, 4 460 mm, compared with a four year mean of 4 000 mm. The number of storms with more than 30 mm rain per hour did not differ before and after treatment. Mean yearly runoff for the four years of measurements were 1910 and 1730 mm for reference catchments W3 and W6, respectively. The response in stream flow is very rapid during rain storms. Discharge may rise from a few liters per second to several hundred within half an hour in a typical event (Fig. 1 and 2).

Suspended load concentrations

Most groups of samples from different watersheds, different periods and from low and high flow samples fitted best log-normal distributions and had equal variances. Therefore the Tukey-Kramer test was used to determine differences in mean suspended load concentrations between watersheds and lf- and hf-samples.

Mean suspended load concentrations during low flows were between

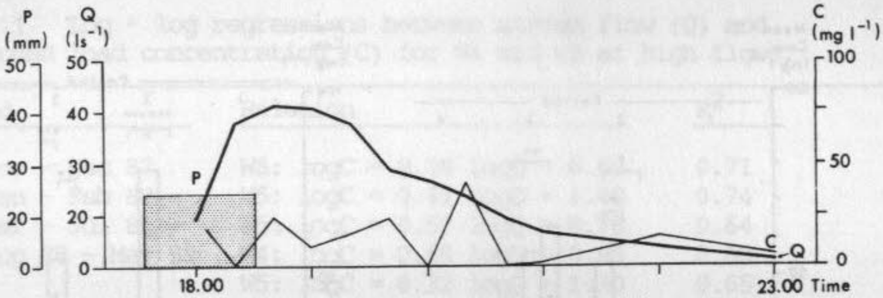

Fig. 1 Typical graph on streamflow (Q) and suspended sediment concentration (C) response on precipitation (P) for reference streams and treated streams before treatment (W6 24 August, 1988).

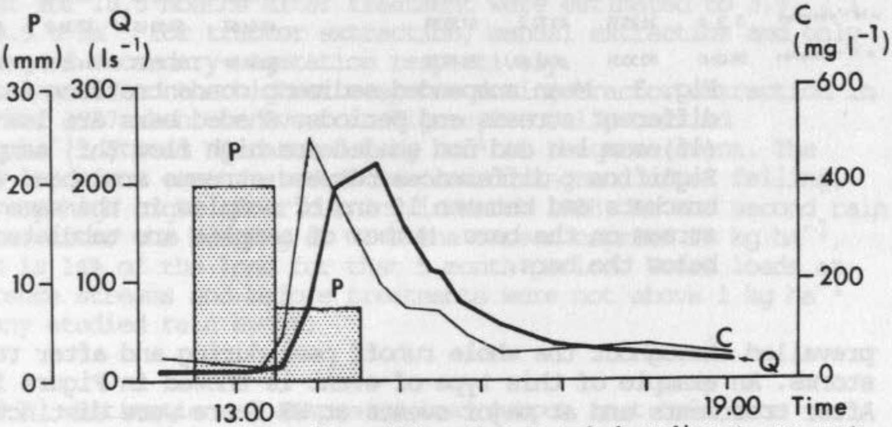

Fig. 2 Streamflow (Q) and suspended sediment concentration (C) response on precipitation (P) for stream W5 during tractor wood extraction (14 December, 1987).

5 to 20 mg l^{-1} for all streams before and after treatments (Fig. 3). There were no significant differences between lf-samples between watersheds for any of the periods monitored. Most mean concentrations during high flows were not significantly different from the lf-samples and between different watersheds (Fig. 3). The most apparent significance in differences in this case are for the tractor treated w5 for all periods after treatment and the manually extracted w4 for the last period after treatment. For the watersheds earlier struck by forest fire, the response to the treatment is not so clear, and both w1 and w3 have mean hf-sample concentrations significantly different from those of lf-samples from the same watersheds. Mean hf-concentrations significantly different from mean lf-concentrations are in all cases, except one, also significantly different from mean hf-concentrations from other streams at the same period, like W5 and W4 relative to the reference stream W6.

Before treatment in all streams and after treatment in reference streams (except a few events in W3), the low concentration normally

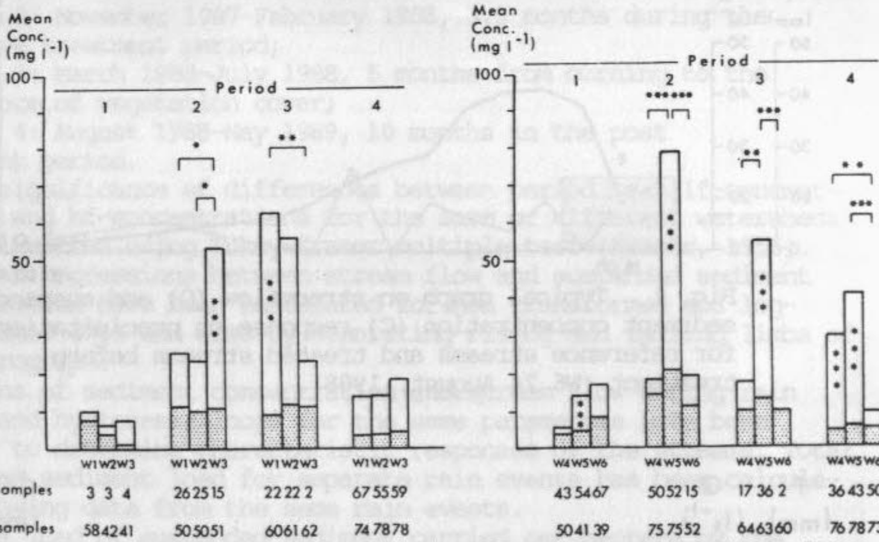

Fig. 3 Mean suspended sediment concentrations for different streams and periods. Shaded bars are low flow (lf) samples and non shaded are high flow (hf) samples. Significant differences between streams are shown with brackets and between lf and hf samples in the same stream on the bars. Number of samples are tabulated below the bars.

prevailed throughout the whole runoff peak during and after rain storms. An example of this type of event is showed in Figure 1. After treatments and at major events at W3 there were distinct peaks of suspended load concentration, like the one in Figure 2. The maximum sampled concentration was 552 mg l^{-1}.

Regressions between streamflow and suspended sediment concentrations were very poor in all cases except during high flows at W5 and W4 after treatment where the mean hf-concentrations were significantly different from the mean lf-concentrations (Table 1). In all other cases the coefficient in the log-linear regression was less than 0.1 and the squared regression coefficient less than 0.5. Tests with regressions performed on differently transformed data or by separating falling and rising limbs of the hydrograph did not give better results.

Before treatment, single storms with random or no response in suspended load concentration were the most common (Fig. 1). After treatment responses became more distinct from all treated watersheds. Single storm events with sediment peak before streamflow maximum like in Figure 2 was most common after treatment in W5, but also single storm events with negative hysteresis loops and with random response have been sampled. No relation depending on rain intensity or soil moisture conditions was found on sediment load response from the limited number of storms sampled in this study.

TABLE 1 Log - log regressions between stream flow (Q) and suspended load concentration (C) for W4 and W5 at high flows.

Period	Relation	R^2
2a. Nov - Dec 87	W5: $\log C = 0.70 \log Q + 0.68$	0.71
2b. Jan - Feb 88	W5: $\log C = 0.41 \log Q + 1.40$	0.74
3. Mar - Jul 88	W5: $\log C = 0.57 \log Q + 0.78$	0.64
4. Aug 88 - May 89	W4: $\log C = 0.66 \log Q + 0.96$	0.68
	W5: $\log C = 0.32 \log Q + 1.30$	0.65

Suspended sediment load

Suspended sediment load output computed for the different watersheds for the different time periods are presented in Table 2. Total output for 18.5 months after treatment were estimated to 3.9, 2.1 and 0.9 t ha^{-1} for tractor extraction, manual extraction and only burning of secondary vegetation respectively.

Stream W5 had the highest response during tractor extraction in December 1987 when the event in Figure 2 totally carried 28 kg ha^{-1}. Stream W4 carried 7 kg ha^{-1} at the same event. The total load from events at W1+2 was low during and after felling, but very high right after burning in March 1988. At the second rain (50 mm) after the burning at W1+2 the stream carried 56 kg ha^{-1}, which is 14% of the load for that 5 month period. Total loads at reference streams and before treatments were not above 1 kg ha^{-1} for any studied rain event.

TABLE 2 Estimation of suspended load output in t ha^{-1} from different watersheds before (period 1) and after treatments (period 2 - 4). Note the different length of periods calculated.

Watershed / Treatment	Period 1	2	3	4
Months	10	3.5	5	10
W 1+2 / secondary, felled, burned and planted	0.1	0.2	0.4	0.5
W 3 / secondary, reference	0.02(a)	0.3	0.2	0.4
W 4 / forest, felled, manual, not burned and planted	0.05	0.1	0.2	1.9
W 5 / forest, felled, tractors burned and planted	0.2	1.0	1.0	1.9
W 6 / forest, reference	0.2	0.1	0.01(a)	0.2

(a) low number of samples causing underestimation (Fig. 4)

DISCUSSION

The analysis of the low concentration of suspended load at low flows was a problem because of the variation of the filter paper weights. This was compensated by a high number of samples. This variation has also caused a higher scatter of concentrations at concentration recession at the end of rain events, resulting in lower significance in regressions between stream flow and concentrations.

The estimated soil loss by suspended load from the catchments was low before treatments and for reference streams. There was also a tendency that the soil loss was larger from the catchments which experienced forest fire in 1982/83 compared to the catchments where no disturbance had ocurred since the selective logging in 1981.

Notable is that the low concentrations and soil loss from the forest, selectively logged 5 - 9 years earlier, are comparable to the figures reported for undisturbed forests (Lal, 1986), despite the fact that the tractor tracks from that logging were still reported to have very low infiltrability (Malmer & Grip, in press). The estimated soil loss might be an underestimation for a longer time perspective, because a great part of soil loss from undisturbed systems has been shown to originate from extreme events (Douglas, 1969) and the period for studies like this one is too short to cover the full variation. It should also be noted that the selective logging performed in the studied forest is a light one compared to what is the standard in many other places or when the same forests are logged several times within a short period.

The suspended load soil loss after treatment was highest after tractors extraction, with a total soil loss of 3.9 t ha^{-1} for 18.5 months after the beginning of treatment. This soil loss was most intensive during log extraction and has decreased as new vegetation covered the ground. Tractor tracks though were still not covered and suspended load concentrations were not back to pretreatment levels after 18.5 months. The high load of 2.3 t ha^{-1} for the same period after manual extraction was not expected. The bulk of this loss (1.9 t) was estimated for the last 10 months of the study period. The reason for this rise, after only a small response to the first storms after treatment, is not obvious. One reason might be that the high amounts of slash left without burning has been suppressing weeds and undergrowth. The growth of the planted trees at this treatment has been much faster in height and volume (unpublished) than with the "normal practice" and the trees very soon closed up to add on the preventive effect of under growth. Then when all the slash starts to decompose, the ground will be left relatively bare and give way to faster runoff during intense rain events. Higher discharge velocities then might cause higher erosion of stream banks and topsoil. The planting rows cleared in the slash were also oriented dowslope which might have caused faster runoff.

The treatment at watersheds W1+2 did not include any soil disturbance from wood extraction, and consequently the soil loss is not very different from the "secondary reference" W3. Notable here is the high soil loss during the storm of 5 March 1988, which was the second rain after the burning of W1+2. The loss of 56 kg ha^{-1}

during this event was 14 per cent of the estimated soil loss for that 5 month period. Before burning (after clearing) and as soon as vegetation has covered the top soil, the concentrations of suspended load were back to the same levels as W3 (Figure 3).

To get an idea of the severity of the estimated soil loss for future sustainable yield from coming land use like forestry, it is essential to discuss the origin and the chemical properties of the particles lost from the basin. For example the particles transported out after burning will carry a high load of nutrients from the burned slash, while sediments eroded from deeper soil at stream banks will mean a less severe loss for future biomass production.

No estimation of the sediment delivery ratio was done in this study. Like in most other cases (Walling, 1988), erosion at spots like tractor tracks was probably considerably larger than what is transported out of the basin. At watershed W5, a considerable amount of erosion has formed gullies in tractor tracks and much of this material has been deposited in lower parts of the watershed with less slope and lower runoff velocities.

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Water-yield changes after clear-felling tropical rainforest and establishment of forest plantation in Sabah, Malaysia

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ABSTRACT

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A paired catchment experiment was conducted in Mendolong, Sabah, Malaysia to monitor water-yield changes due to different methods of clear-felling tropical rainforest and establishment of fast plantation with fast-growing trees (*Acacia mangium*). The study included five catchments; treatments before planting were: (1) cutting and burning of secondary vegetation (forest fire 1982/1983), (2) clear-felling, tractor log extraction and no burning, (3) clear-felling, tractor log extraction and burning. Two catchments were monitored as controls: one for the secondary vegetation and one for the rain forest. A calibration monitoring period for all catchments for 27.5 months started in August 1985. The results presented here include the calibration period and 32.5 months during and after treatments. Multiple regression analyses on runoff from treated and reference catchments before and after treatment were used to determine the increase in runoff due to treatments.

Mean yearly areal rainfall was 3352 mm and mean yearly runoff 1950 mm for the control catchments for the 5 years of study. Calculated water-yield increases were for (1) 1000, (2) 447 and (3) 1290 mm for the first 32.5 months during treatment and plantation establishment.

The fastest runoff generation during storms was found after cutting, burning and no soil disturbance, leaving no vegetation to transpire and minimum disturbance to natural waterways. The use of tractors, causing soil disturbance, severe loss of infiltrability and top-soil hydraulic conductivity resulted in decreased storm stormflow discharge during and after treatment. Later, with maintained low infiltrability and prolonged erosion clearing new waterways instead of the disturbed ones, a faster runoff generation and larger runoff increase were found in the last year of study.

INTRODUCTION

The effect of forest cover removal on water yield has been reviewed by Bosch and Hewlett (1982); the increase in water yield is well documented, but mostly from temperate climates. During recent years the evidence for increase in water yield following removal of tropical forests has accumulated

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(Bruijnzeel, 1990). However, comparisons of the hydrological consequences of different forestry treatments including the establishment of forest plantations are still lacking, both from South East Asia and other parts of the tropics.

The present paired catchment study, started in 1985, monitors the hydrological, hydrochemical, soil and biomass changes before, during and after different ways of converting selectively logged tropical rainforest and forest struck by forest fire to forest plantation. This paper presents the changes in water yield for the first 32.5 month after treatment compared with a 27.5 month calibration period.

RESEARCH AREA

The Mendolong research area which includes the six catchments used for this impact study is situated at 650–750 m a.s.l. on the foothills of Mount Lumako in the Crocker Range 35 km southeast of the coast at Sipitang (115.5°E, 5.0°N) in Sabah, Malaysia. The natural vegetation of the research area is a lowland hill dipterocarp forest (Whitmore, 1984, after Symington, 1943), although the altitude is just below the transition to lower montane forest. The size of the catchments varies from 3.4 to 18.2 ha, and the treatments cover 100% of the basins studied. Slopes in the area are mostly moderate, ranging up to 27%, but steeper slopes of up to 57% also occur. The drainage pattern of the catchments can be seen in Fig. 1.

The nature of some soil physical properties in the area has been reported by Malmer and Grip (1990). They reported a decrease of mean steady-state infiltrability on clay soil from 154 to 0.3 mm h^{-1} on tractor tracks and to 36.7 mm h^{-1} on areas disturbed by manual log extraction. Andersson (1990) also found a reduction from 0.68 to 0.032 mm h^{-1} of top soil field saturated hydraulic conductivity (20 cm depth) on distributed clay soil.

The rainfall in the trial site is mostly convective, but there is also a monsoon influence (Wong, 1989).

The research area includes two main types of vegetation. Three catchments (W1–W3) comprised forest, selectively logged in 1981 and later struck by forest fire in 1982–1983 (Beaman et al., 1985; Malingreau et al., 1985). The remaining catchments (W4–W6) comprised dipterocarp forest lightly selectively logged in 1981.

SUCCESSION OF VEGETATION

Biomass dry weight estimations based on destructive sampling before and after treatments were reported by Sim and Nykvist (1991). The hydrological monitoring started in 1985, 2.5 years after the forest fire. At this time the

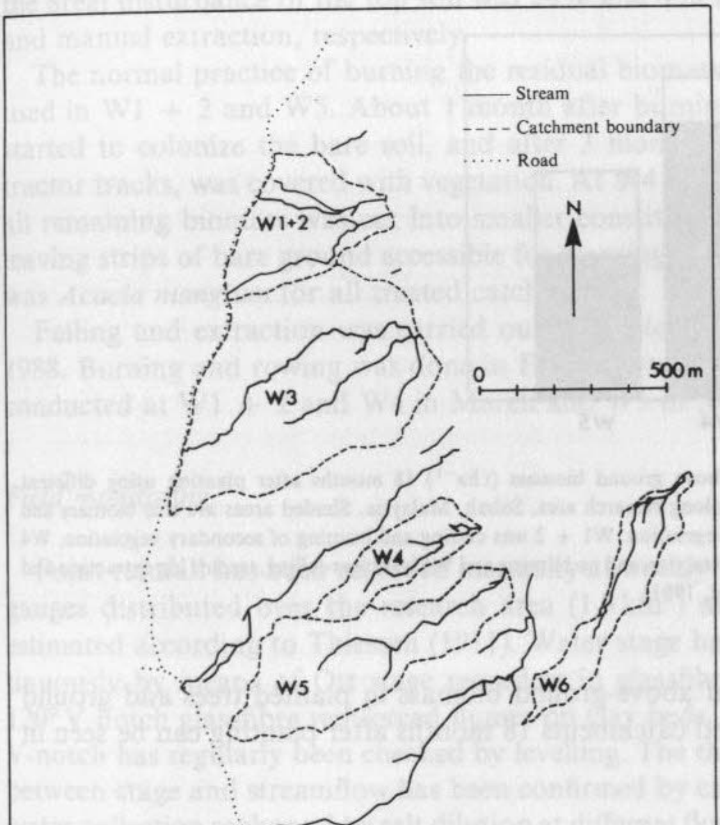

Fig. 1. Map of catchments W1 + 2–W6 in Mendolong research area, Sabah, Malaysia.

biomass at W1 + 2 was only 4.7 t ha^{-1}, mostly comprising ferns, grasses and herbs. In late 1987, just before treatment, the biomass had increased to 26.7 t ha^{-1} of which 20.2 t ha^{-1} was accumulated in small pioneer trees. Standing and lying dead trees are not included in these figures. Biomass in W3 and W6 was not estimated. However, prefire vegetation in W3 was not as damaged by the forest fire as W1 + 2. The areal extent of the forest fire in W3 was 80.4%, and, in a gradient from the water divide between W2 and W3 to where the fire stopped, there were more surviving trees from the prefire period and less pioneer vegetation.

Above-ground biomass in W4 and W5 before treatment was 276 and 245 t ha^{-1}, respectively, at the time of clear-felling.

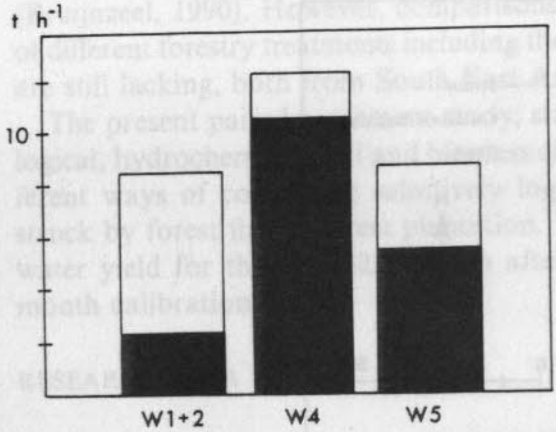

Fig. 2. Total dry weight of above ground biomass ($t\,ha^{-1}$) 18 months after planting using different preplanting methods in Mendolong research area, Sabah, Malaysia. Shaded areas are tree biomass and non-shaded areas are ground vegetation. W1 + 2 was cutting and burning of secondary vegetation, W4 was clear-felling, manual log extraction and no burning and W5 was clear-felling, tractor log extraction and burning (after Sim and Nykvist, 1991).

The accumulation of above-ground biomass in planted trees and ground vegetation in the treated catchments 18 months after planting can be seen in Fig. 2.

MATERIALS AND METHODS

Forestry treatments

The treatments, covering 100% of the catchments, were as follows:

W1 Non-mechanized clearing of remaining trees and secondary vegetation, no wood extraction, burning of all biomass and planting.

W2 The same treatment as W1.

W3 Control catchment for forest fire area. No treatment.

W4 Manual felling, manual wood extraction, clearing of planting rows in the slash and planting these rows, without burning.

W5 Manual felling, wood extraction using crawler tractors, burning of the remaining biomass and planting (normal practice).

W6 Control catchment for selectively logged area. No treatments.

Catchments W1 and W2 were treated as one because of uncertainty in identifying the phreatic water divide between the two streams. Malmer and Grip (1990) described the wood extraction methods used in this study in detail and the resulting effects on physical properties of the soil. They reported that the areal disturbance of the top soil was 24% and 4% for tractor extraction and manual extraction, respectively.

The normal practice of burning the residual biomass before planting was used in W1 + 2 and W5. About 1 month after burning, grasses and weeds started to colonize the bare soil, and after 3 months most soil, except for tractor tracks, was covered with vegetation. At W4 burning was avoided and all remaining biomass was cut into smaller constituents and put into rows, leaving strips of bare ground accessible for planting. The planted tree species was *Acacia mangium* for all treated catchments.

Felling and extraction was carried out from November 1987 to January 1988. Burning and rowing was done in February to April, and planting was conducted at W1 + 2 and W4 in March and W5 in May 1988.

Field monitoring

Point rainfall has been recorded manually at weekly intervals with 12 rain gauges distributed over the research area ($1.9\,km^2$) and areal rainfall was estimated according to Thiessen (1911). Water stage has been recorded continuously by means of Ott stage recorders in glassfibre wells connected to 120° V-notch glassfibre reinforced flumes on clay beds. The base level of the V-notch has regularly been checked by levelling. The theoretical relationship between stage and streamflow has been confirmed by calibrations both using water collection sacks and by salt dilution at different flows and different times at all flumes.

Deep leakage of ground water under the flumes was, under normal circumstances, negligible due to the shallow soils (1–3 m) and to a rapidly increasing clay content with depth (Malmer and Grip, 1990) causing low hydraulic conductivity except for the top soil (Andersson, 1990). However, at W4, a constant leakage of some 250 mm $year^{-1}$ began in March 1987. At W4 the protective clay under the flume is underlain by soil of high sand content. A protective canvas dug down upstream of the flume was damaged at a point where the protective layer of clay was inadequate.

Data analysis

Owing to the small size and fast runoff response of the catchments, water stage was digitalized using 10-min means to calculate mean values of weekly streamflow and runoff. Periods of missing streamflow data, due to malfunction of the stage recorders, have been replaced with data from regression models (based on weekly data to avoid serial correlation) using data on areal

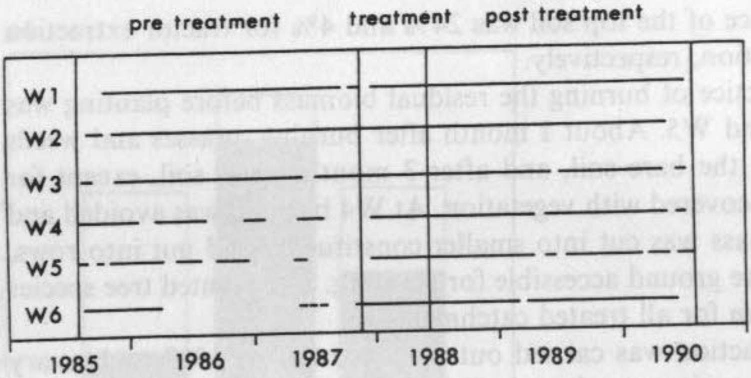

Fig. 3. Periods of measured water stage data from catchments in Mendolong research area, Sabah, Malaysia. Periods of missing data, due to malfunctioning of recorders, are indicated by discontinuities; the missing data have been replaced using regression models on streamflow between different streams.

rainfall and streamflow from other streams. Periods of measured and calculated data can be seen in Fig. 3.

The effects of treatments on runoff during different periods were detected and tested by the use of dummy variables in regressions between treated streams and reference streams (Gujarati, 1978). The periods compared were:

(1) calibration period August 1985–November 1987;
(2) treatment period November 1987–July 1988;
(3) first hydrological post-treatment year August 1988–July 1989;
(4) second hydrological post-treatment year August 1989–July 1990.

All four periods were treated in one regression of the general form

$$R_t = a_1 + a_2 R_r + a_3 P_t + a_4 D_1 + a_5 D_2 + a_6 D_3 + a_7(D_1 R_r) + a_8(D_2 R_r) + a_9(D_3 R_r) + a_{10}(D_1 P_t) + a_{11}(D_2 P_t) + a_{12}(D_3 P_t); \; D_n = (0, 1) \quad (1)$$

where R_t is predicted weekly runoff from a treated basin and the predictors are R_r (weekly runoff from control basin) and P_t (weekly precipitation on the treated basin). The dummy variables D_n all equal 0 during period 1 (pretreatment) and $D_1 = 1$ only during period 2, $D_2 = 1$ only during period 3 and $D_3 = 1$ only during period 4. By testing if every coefficient a_n differed significantly from zero, every difference in slope and intercept between the four periods was detected and tested as recommended by Hewlett et al. (1984).

To estimate the actual difference in runoff caused by the treatments, the regression for period 1 was used to model the runoff from the treated basins as if there had been no treatments.

The adjusted R^2 denotes the proportion of variance in the dependent variable expected to be accounted for by the model when taking a new sample from the same population. It has the form

$$1 - (1 - R^2)(N - 1)/(N - p) \quad (2)$$

where N is number of cases and p is the number of predictors, including the intercept constant.

When measurements started in 1985, catchments W1 + 2 were still under recession from the increased runoff which resulted from the forest fire in 1982/1983. This was not the case for the control catchment W3, so that the relation between these catchments was not constant throughout the calibration period. This was to some extent compensated for by not using the first year of calibration to calculate the regression relationship between these streams.

To compensate for the leakage after March 1987 the measured weekly runoff data for W4 (R_m) were replaced by data from a regression model. This model derived by expressing values of W4 (R_e) as a function of runoff in W6 and W3 for the time before leakage appeared in the pretreatment period (August 1985–February 1987).

$$R_e = 0.366R_{w6} + 0.617R_{w3} \ (n = 82, \text{adjusted } R^2 = 0.880) \quad (3)$$

Values of R_e obtained by applying relationship (3) were used as dependent variables in a regression with values of runoff actually measured for W4 as an independent variable during the period of leakage but before treatment (March 1987–November 1987).

$$R_e = 4.877 + 1.071R_m \ (n = 37, \text{adjusted } R^2 = 0.928) \quad (4)$$

By assuming the model of the leakage (eqn. (3)) to be valid also after treatment, a model for actual runoff for W4 that compensated for leakage (R_a) could be achieved by replacing R_e with R_a and R_m with R_e in eqn. (4). Note that the slope coefficient in eqn. (4) gives evidence to the constant nature of the leakage during the period before treatment.

To detect changes in flow regimes of the treated streams an analysis of flow dynamics was done based on 2 h mean values of specific discharge ($l\,s^{-1}\,ha^{-1}$). As for weekly data above, regression models were used to make a complete data set of 2 h discharge for the six streams for the 5 years of measurement. In this case data for both actual time (t) and adjacent time ($t - 1$) were used with the predictors in the models to account for serial correlation. Baseflow and stormflow separation were done using straight line separation of the hydrographs. Mean specific discharges for baseflow, stormflow and flowpeak values were compared between different time periods ((1)–(4) above) and different streams using the Tukey–Kramer multiple test (Kramer, 1956).

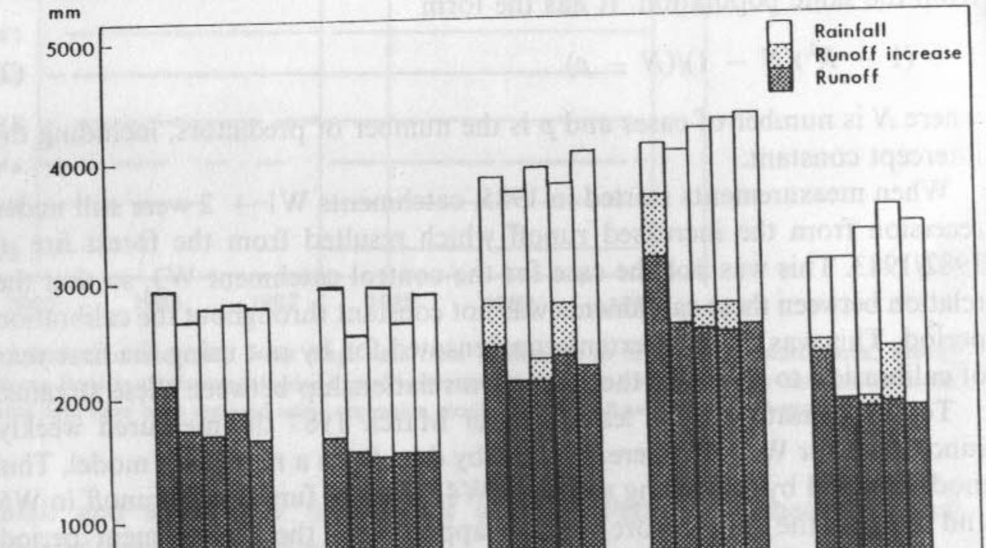

Fig. 4. Rainfall and part of that occurring as runoff and runoff increases due to treatment (mm) for 5 hydrological years for catchments in Mendolong research area, Sabah, Malaysia, before and after different forestry treatments.

RESULTS

Rainfall

Yearly areal rainfall for the different catchments is presented in Fig. 4. Mean yearly areal rainfall was 3215 mm and 3490 mm for control catchments W3 and W6, respectively, for the 5 years of this study. Mean yearly point rainfall was 3341 mm for the 5 years of study. Mean yearly point rainfall for the whole research area for the 2 years before treatment (2647 mm) was lower than for the first 3 years during and after treatment (3804 mm), of which the first year after treatments 1988/1989 had the greatest mean point rainfall with 4113 mm. The greatest difference in point rainfall within the research area for 1 year was 928 mm in 1986/87. The average number of days with rain per year was 143 for the 5 years of data presented here. Most rain events are of short duration with highest intensity during the first 30 min. Maximum weekly point rainfall was 340 mm. The longest period without rain was 17 days, in

December–January 1989/1990 and again in June–July 1990. Periods without rain for more than 10 days occur regularly around July and December.

Runoff

Yearly data on runoff from the catchments can be seen in Fig. 4. Mean yearly runoff for the 5 years was 1962 mm and 1950 mm for control catchments W3 and W6, respectively. Maximum recorded weekly runoff for control catchments was 159 mm (W6) and for treated catchments 228 mm (W1 + 2). During the regular dry spells described above the smallest catchments (W1 + 2, W4 and W6) sometimes had no runoff for a few days, but these periods have never exceeded 1 week.

As a comparison, data on runoff were also calculated from the 2 h data set used for flow dynamic analysis. Mean yearly runoff for the 5 years differed by only a few millimetres for control catchments W3 and W6 (1963 and 1955 mm, respectively). The maximum deviation in these calculations is for W4 which shows 431 mm less runoff during the 5 years of study, even after the corresponding correction for the leakage made on the weekly data.

Water-yield changes

The significantly different regression equations between runoff from treated catchments and from control catchments, and areal rainfall during periods (1)–(4) are presented in Table 1.

Calculated yearly changes in runoff due to treatments in W1 + 2, W4 and W5 are tabulated in Table 2. Note that only 8.5 months of the hydrological year 1987/1988 (first year of treatment in Fig. 4) account for the period of treatment and post-treatment, while the division between periods (1) and (2) used for regression analysis was in November 1987 when the actual treatments started.

Discharge dynamics

The runoff from the catchments occurs mainly in stormflows. The percentage of baseflow over 5 years were 30% and 26% for reference catchments W3 and W6, respectively.

Mean baseflow discharge for different periods and for different streams with significant differences between treated and control streams is shown in Fig. 5(a). During the calibration period both control streams W3 and W6 had significantly lower mean baseflow discharges than W1 + 2, and W4 and W5, respectively. On the other hand, after treatment, W1 + 2 and W4 had the same or significantly lower mean baseflow discharges than the control

TABLE 1

Regression equations between runoff from treated catchments and that from control catchments (R_{wn}) and catchment areal rainfall (P_{wn}) at Mendolong research area, Sabah, Malaysia

Relation	Significance[a]	N (weeks)	Adj R^2
Period (1)			
$R_{w1+2} = 1.110R_{w3}$		67	0.924
$R_{w4} = 0.670R_{w6} + 0.107P_{w4} + 2.589$		119	0.909
$R_{w5} = 0.661R_{w6} + 0.224P_{w5} - 2.737$		119	0.922
Period (2)			
$R_{w1+2} = 1.110R_{w3} + 0.210P_{w1+2}$	***	37	0.924
$R_{w4} = 0.670R_{w6} + 0.107P_{w4} + 7.792$	**	37	0.909
$R_{w5} = 0.661R_{w6} + 0.224P_{w5} + 9.733$	***	37	0.922
Period (3)			
$R_{w1+2} = 1.110R_{w3} + 0.473P_{w1+2} - 21.459$	***	52	0.924
$R_{w4} = 0.670R_{w6} + 0.107P_{w4} + 10.340$	***	52	0.909
$R_{w5} = 0.661R_{w6} + 0.224P_{w5} + 2.289$	**	52	0.922
Period (4)			
$R_{w1+2} = 1.110R_{w3} + 0.265P_{w1+2} - 9.844$	***	52	0.924
$R_{w4} = 0.670R_{w6} + 0.153P_{w4} + 2.589$	*	52	0.909
$R_{w5} = 0.661R_{w6} + 0.343P_{w5} - 2.737$	***	52	0.922

[a] Degree of significance for how the difference between coefficients in periods after treatment and the same coefficients in the calibration period equation for respective catchment differ from 0.

Period (1) was the 27.5 months calibration period, period (2) was 8.5 months during and after treatments and periods (3) and (4) were the first and second hydrological years after treatment, respectively.

TABLE 2

Increases in runoff (mm) from different catchments due to different forestry treatments in Mendlong research area, Sabah, Malaysia

Catchment	First year	Second year	Third year	Total 3 years
W1 + 2	397	522	89	1008
W4	197	170	80	447
W5	460	262	468	1190

W1 + 2 was cutting and burning of secondary vegetation, W4 was clear-felling, manual log extraction and no burning, and W5 was clear-felling, tractor log extraction and burning, before planting of fast-growing trees.

streams, indicating a relative lowering of baseflows after treatments. W5 still had a higher mean baseflow discharge than W6 during periods (2) and (3) and did not show less mean baseflow discharge than W6 until period (4).

Mean stormflow discharges for different periods and different streams with significant differences between treated and control streams are shown in Fig. 5(b). During the reference period there were no significant differences in mean stormflow discharges. For W1 + 2 there was a significant increase compared with W3 for all periods after treatment. W4 showed no differences compared with W6, while W5 showed a significant decrease in mean stormflow discharge only in period (3) compared with W6.

Mean peak specific discharges for different periods and different streams with significant differences between treated and control streams are shown in Fig. 4(c). Like mean stormflow discharge, there were no significant differences in mean peak discharge during the calibration period. The only significant increase after treatments was for W1 + 2 which had markedly higher peak discharges than W3 during all three periods after treatment.

DISCUSSION

General remarks

At periods of measured runoff and no malfunction (Fig. 2) the maximum error in weekly runoff is estimated to be about 5%. During periods when data were missing and values were calculated, the error might rise to a maximum of 10%; this error is less than 200 mm $year^{-1}$ in the worst possible case. For the regression analysis, measurement errors are assumed to equal zero over a long period of study.

All the water-yield changes for the treated catchments are significant as compared with the extrapolation of the relations between treated and control streams in the calibration period. The calculated increases (1008 mm, 447 mm and 1190 mm for 32.5 months for W1 + 2, W4 and W5, respectively) were all greater than the maximum possible measurement and calculation error, except for W4 which is slightly below.

Rainfall

The rainfall at the research area is relatively high and has a high variability. There seems to be a local maximum of rainfall at the research area. Rainfall in the Mendolong catchment area is higher than rainfall at the surrounding meteorological stations in tree plantations at both higher and lower elevations (Wong, 1989). Walsh (1982a) discussed the possibility of a similar maximum

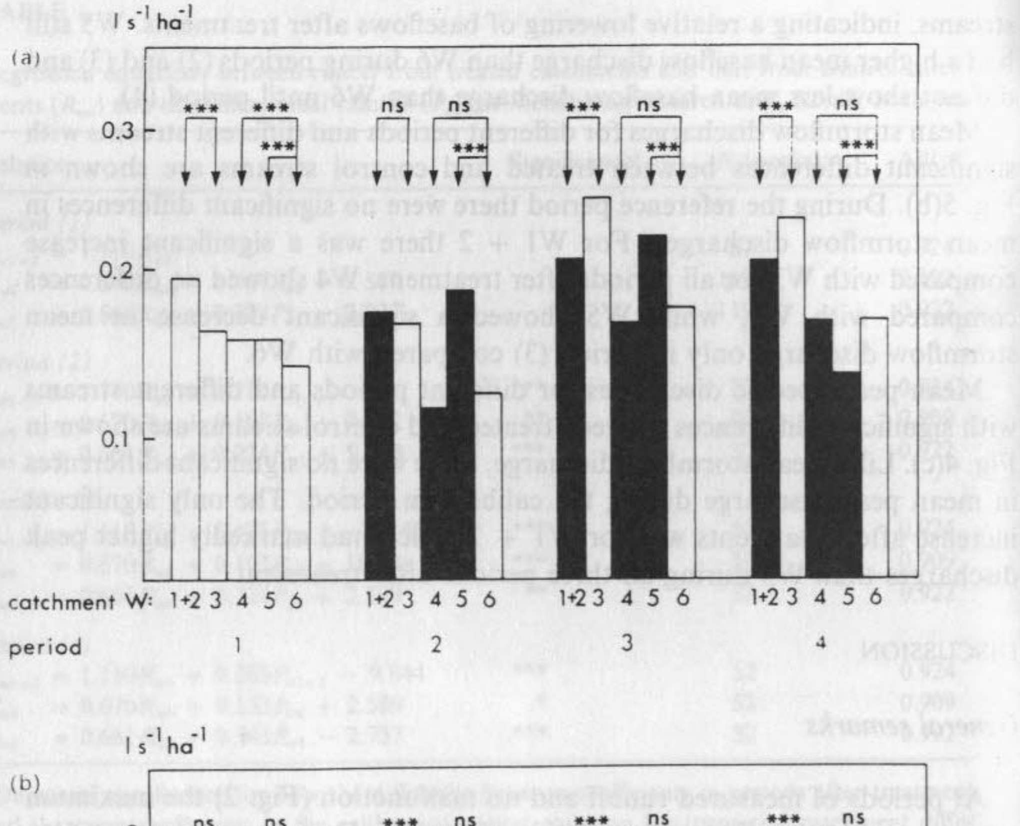

Fig. 5. (a) Mean baseflow specific discharge, (b) mean stormflow specific discharge, and (c) mean peak specific discharge ($l\,s^{-1}\,ha^{-1}$) for different periods for catchments in Mendolong research area, Sabah, Malaysia. ***, significance of differences between treated and control catchments during different periods. Period (1) was 27.5 months before treatment, period (2) 8.5 months treatment period and periods (3) and (4) were the first and second hydrological years after treatment, respectively.

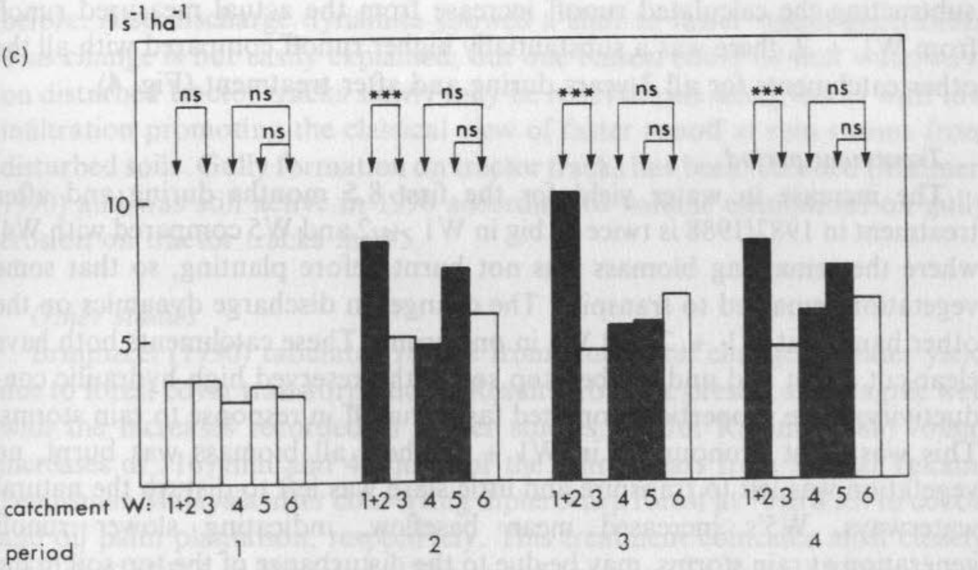

Fig. 5. Continued.

caused by orographic uplift at Gunung Mulu National Park in northeastern Sarawak, and Lockwood (1974) considered the same subject for Peninsular Malaysia and Central Java.

Water-yield and discharge dynamics

At clear-felling, transpiration is drastically decreased. Evaporation will also decrease, as the aerodynamic resistance increases compared with that over the forest canopy. As the ground vegetation and the planted trees develop, the leaf area and transpiration will return to pretreatment levels, while the aerodynamic resistance will still be rather high as long as the stand height is small.

Changes in discharge will reflect alterations in the water use of the vegetation and soil disturbance. The soil physical properties indicate a situation for natural runoff generation close to that described by Elsenbeer and Cassel (1990) with near-surface and surfical waterways contributing to a rapid runoff response to rainstorms. Also, the main part of the runoff occurs as stormflow.

During the calibration period catchments W1 + 2 were still under a greater recession from the increased runoff after the forest fire 2.5 years earlier, than was control catchment W3 (Fig. 3). The adjustment, made to compensate for that, increased the calculated change in water yield for W1 + 2 by about 100 mm year^{-1} from 712 to 1008 mm for the 32.5 months. This runoff increase for W1 + 2 might still be an underestimation. Even after subtracting the calculated runoff increase from the actual measured runoff from W1 + 2, there was a substantially higher runoff compared with all the other catchments for all 3 years during and after treatment (Fig. 4).

Treatment period

The increase in water yield for the first 8.5 months during and after treatment in 1987/1988 is twice as big in W1 + 2 and W5 compared with W4, where the remaining biomass was not burnt before planting, so that some vegetation remained to transpire. The changes in discharge dynamics on the other hand, put W1 + 2 and W4 in one group. These catchments both have clear-cut forest and undisturbed top soil with preserved high hydraulic conductivity; these properties promoted faster runoff in response to rain storms. This was most pronounced in W1 + 2 where all biomass was burnt, no vegetation was left to transpire and little slash was left to disturb the natural waterways. W5's increased mean baseflow, indicating slower runoff generation at rain storms, may be due to the disturbance of the top soil in the tractor tracks. The infiltration was drastically reduced causing overland flow and ponding, but all water may not reach the stream at once. Disturbance of the topsoil also obstructs normal waterways and water may be trapped in ponds in the tracks for several days (Malmer and Grip, 1990), especially during the treatment period. Uphill logging (Megahan and Schweithelm, 1983) was used; this does not concentrate surface runoff downslope along the tractor tracks as does downhill logging, with the landing site in the lowest part of the logged area. This suggests that estimated data on suspended sediment load due to tractor extraction (Malmer, 1990) may be low compared with logging operations using downhill logging or 'timber cruising').

Post-treatment period

The differences in the increases in water yield between the treated catchments during 1988/1989 can be seen in terms of regrowth of biomass as ground vegetation and planted trees (Fig. 2). W4, which had the fastest growth of trees, had the smallest increase in runoff. W5 had about the same biomass as W1 + 2, but better growth of planted trees; consequently W1 + 2 had the greatest runoff increase. The discharge dynamics during 1988/1989 for the treated basins compared with the controls were very similar to the first treatment period discussed above.

During 1989/1990 the leaf areas in the catchments were substantially increased; the canopy also closed, except on the tractor tracks in W5 and on patches with very bad tree growth in W1 + 2. This increase in biomass and water use explain the reduced increase in runoff for W1 + 2 and W4. The increase in runoff from W5 on the other hand, was greater than that the year before. Also discharge dynamics showed a shift to faster runoff generation. This change is not easily explained; but one reason could be that waterways on disturbed tractor tracks slowly may be redeveloped along tracks with low infiltration promoting the classical view of faster runoff at rain storms from disturbed soils. Gully formation on tractor tracks has been recorded (Malmer, 1990) and was still active in 1990 according to volume estimations on gully erosion on tractor tracks in W5.

Other studies

Bruijnzeel (1990) tabulated results from studies on change in water yield due to forest cover transformations. Results from the present study agree well with the increases recorded in earlier studies. Abdul Rahim (1988) found increases of 1169 mm and 437 mm for the 3 first years from Sungai Tekam, Peninsular Malaysia after converting dipterocarp forest at 70 m a.s.l. to cocoa and oil palm plantation, respectively. This treatment coincides most closely with that of W5 in this study. The runoff increases in Sungai Tekam were higher compared with this study in terms of the increase for the first year of treatment, and very much higher in terms of increase in percentage of pretreatment runoff. Comparing the discharge dynamics, Sungai Tekam also shows an increase of the baseflow component like W5 and a markedly faster runoff at rain storms after some time and especially after clearing the stream of logs disturbing the waterway (Bruijnzeel, 1989a, after Drainage and Irrigation Department (DID), 1986).

Evapotranspiration

Mean evapotranspiration, as calculated from the water balance for catchments W3 and W6 for the 5 years studied, was 1253 mm and 1540 mm, respectively. This difference is most probably related to the mature forest at W6 compared with the mix of surviving trees and secondary vegetation in W3. Compared with the review by Bruijnzeel (1989b) on evapotranspiration of tropical forests, the dipterocarp forest in Mendolong had evapotranspiration slightly above a mean value for Southeast Asian tropical lowland forests. Walsh (1982b) reported a water-balance-derived evapotranspiration of 1743 mm from the Melinau catchment in northeastern Sarawak (only 120 km southwest from Mendolong and with similar topography and high rainfall). The Melinau catchment is partly underlain with limestone making an overestimation due to leakage possible. The small size of the catchments in this study and the technique with undisturbed soil under the flumes increases the risk of leaching error in the water balance. On the other hand, relatively shallow soils of low hydraulic conductivity, except for the top soil (Andersson, 1990), suggest that substantial deep leakage is unlikely. If deep leakage occurs, the calculated values of evapotranspiration are proportionally overestimated, but the measured effects of treatments on the increase in water yield are still valid as descriptions of what happens in the upper soil horizons.

For the 3 years after establishment of the plantation, yearly mean evapotranspiration was 719 mm, 1684 mm and 1247 mm for W1 + 2, W4 and W5, respectively, due to the very different regrowth from different treatments. Wong (1989) estimated mean yearly potential evapotranspiration 1986–1988 for similar tree plantations in the area as 1573 mm, based on Class A evaporation pan data from similar tree plantations in Tawau, southeastern Sabah.

The regular periods of no rainfall for up to 17 days reported here raise the question of the ecological significance of drought in this region. This subject has been reviewed by Whitmore (1984). Wong (1989), using weekly data on potential evapotranspiration and rainfall, reported a maximum of 13 weeks with more than 50 mm potential soil–moisture deficit during the 3 years (1986–1988) for stations surrounding the Mendolong catchments. The longest period with this deficit was 5 weeks. This might be a problem accentuated in the planting phase and especially on tractor tracks where compacted soil depresses root growth and evaporation is promoted by the lack of ground vegetation and a litter layer.

CONCLUSIONS

The clearing of vegetation is the main reason for increased runoff. Burning of slash after clear-felling adds substantially to the increased water yield as even less vegetation is left to transpire. Soil disturbance and loss of infiltrability does not in itself substantially increase runoff as compared with minimum soil disturbance and burning in the first stage; it did not even contribute to faster runoff and to an increase of the stormflow component, in the first treatment period. But for the total period after treatment, the water yield from W5 is the highest with both soil disturbance and burning. Also, the increased water yield from W5 might well pertain to a long time as new waterways develop along the tractor tracks; the rehabilitation of disturbed soil has been shown to be very slow. In contrast, using no burning and minimum soil disturbance resulted in a more than 50% reduction in water-yield increases for the first 32.5 months after treatment.

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CONVERTING TROPICAL RAINFOREST TO FOREST PLANTATION IN SABAH, MALAYSIA.
I. DYNAMICS AND NET LOSSES OF NUTRIENTS IN CONTROL CATCHMENT STREAMS.

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IV

ABSTRACT

Streamwater chemistry was monitored during five years in six streams in a paired catchment experiment in Mendolong, Sabah, Malaysia, including controls in rain forest and secondary vegetation after the "Borneo fire" 1982/83 and comparing effects of different ways to establish forest plantations with *Acacia mangium*. Three catchments were covered with selectively logged lowland hill dipterocarp forest (W4 - W6) and three (W1 - W3) with secondary vegetation after forest fire. The control catchments, W3 and W6 covered in this paper, had no treatments applied. Reference monitoring in all streams were 20 months and the total period of study reported here is 64 months. The soil type of the catchments were mainly Orthic Acrisol in W3 and Gleyic Podzol in W6 and a mix of both soil types in the other catchments.

Element baseflow concentrations were generally low and not significantly different from stormflow concentrations for all streams during the reference period. Also were concentrations generally consistent and low for the two control streams during the whole period of measurements. Chemical input as wet deposition was low, due to the large part of rain emanating from local convection.

The rain forest on the Podzol had a tight nutrient circulation indicated by small net losses of macro nutrients. The Podzol was also concluded to have worse circumstances for soil mineralization and more superficial runoff, resulting in higher loads of S, C and N in organic phases, with higher organic C/N ratio, in the discharge. N was found to accumulate in both catchments. An almost double N accumulation in W3 was attributed to larger biomass accumulation still going on after the forest fire 3 - 8 years earlier. On the other hand the Acrisol in W3 had much larger net losses of S, Si, K, Ca, Mg and Na. Most of this difference could be attributed to differences in weathering between the soils and local hydrological differences.

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INTRODUCTION

The last decades have seen an increased trend towards plantation forestry and extractive tree crops in the tropics (Whitmore, 1984). The increment of areas of forest plantations can also be expected to continue ahead in the future (Evans, 1982; Whitmore, 1990). The evaluation of sustainable productivity in terms of nutrients for a tree plantation is complex, and especially data on hydrological nutrient losses and rates of chemical weathering are scarce from tropical forests and plantation establishment (Bruijnzeel, 1990; Anderson and Spencer, 1991).

The present paired catchment study, started in 1985, is monitoring the hydrological, hydrochemical, soil and biomass changes before, during and after different ways of converting selectively logged tropical rainforest and forest struck by forest fire to forest plantation. This paper presents data on dynamics of dissolved elements and their discharge in streamwater leaving the control catchments during a five year period, as well as the catchment net loss or gain of nutrients after subtracting wet deposition. Also some data from the other catchments during the reference period are used for comparison.

RESEARCH AREA

The Mendolong research area including the six catchments used for this impact study is situated at 650 - 750 m.a.s.l. on the foothills of Mount Lumako in the Crocker Range 35 km south east of the east coast of Sabah, Malaysia at Sipitang (115.5° E, 5.0° N). The natural forest of the research area is a lowland hill dipterocarp forest (Whitmore, 1984, after Symington, 1943), although the altitude is just below the transition to lower montane forest. The size of the control catchments W3 and W6 are 18.2 and 4.5 ha respectively.

Slopes in the area are mostly moderate, ranging up to 27%, but steeper slopes of up to 57% also occur. The research area includes two different soil types, Orthic Acrisol and Gleyic Podsol (Figure 1). Control catchment W3 is wholly dominated by Orthic Acrisol, while W6 is mainly consisting of the Gleyic Podsol. Intermediate soil profiles in transitions between the two soil types occur, especially in the areas close to the boundary between the two main areal distributions. The Podsol areas are located where the topography are more gentle and top soils more sandy. The nature of some soil physical properties in the area has been reported by Malmer and Grip (1990). The bedrock of the research area is sedimentary and belongs to the Crocker formation of interbeded sandstone, mudstone and shale (Acres et al., 1975). The interbeded nature of the bedrock makes very local differences possible. Bedload material carried by the streams indicate that shales are more abundant in W3 (Malmer, unpublished).

The rainfall at the trial site is mostly of convective origin, but there is also a weak influence of monsoonal rain (Wong, 1989). Water balance and water discharge dynamics for the catchments 1985 - 1990 has been reported by Malmer (1992). Mean yearly areal rainfall was 3215 mm and 3490 mm for W3 and W6 respectively during the five year period. For the same period mean yearly catchment derived evapotranspiration was 1253 mm (W3) and 1540 mm (W6). The most humid year was 1988/89, the first year after treatments, with 4057 mm and 4340 mm areal rainfall for W3 and W6 respectively.

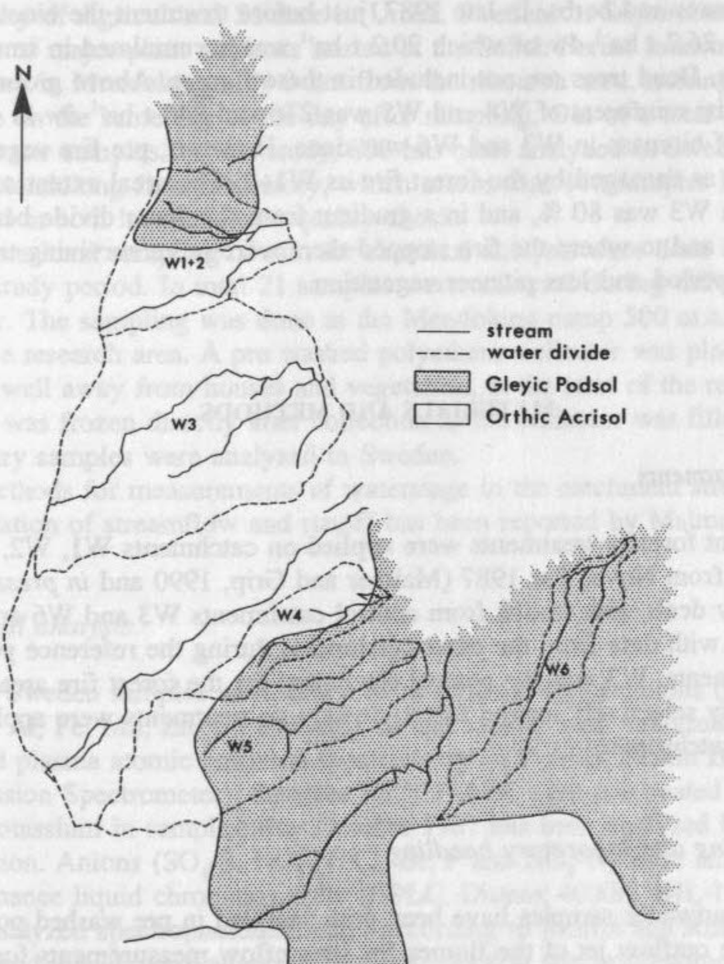

Figure 1. *Map of the Mendolong research area showing and catchments, W1-W6, with streams and soil types (Orthic Acrisol and Gleyic Podsol). Podsol area shaded.*

The research area includes two main types of vegetation. Three catchments (W1 - W3) comprised of forest selectively logged in 1981 and later struck by forest fire in 1982/83 (Woods, 1989). The remaining catchments (W4 - W6) comprised of dipterocarp forest lightly selectively logged in 1981 (Dbh distribution in Sim and Nykvist, 1991).

Biomass dry weight and nutrient contents based on destructive sampling before and after treatments were reported by Sim and Nykvist (1991). The hydrochemical monitoring started in November 1985, 2.5 years after the forest fire. At this time the biomass at W1+2 was only 4.7 t ha^{-1} dw, mostly comprised of ferns, grasses and herbs. In late 1987, just before treatment the biomass had increased to 26.7 t ha^{-1} dw of which 20.2 t ha^{-1} was accumulated in small pioneer trees. Dead trees are not included in these figures. Above ground biomass in the rainforest of W4 and W5 was 276 and 245 t ha^{-1} dw in 1987. No estimation of biomass in W3 and W6 was done. However, pre fire vegetation in W3 was not as damaged by the forest fire as W1+2. The areal extent of the forest fire in W3 was 80 %, and in a gradient from the water divide between W2 and W3 and to where the fire stopped there were more surviving trees from the pre fire period and less pioneer vegetation.

MATERIALS AND METHODS

Forestry treatments

Different forestry treatments were applied on catchments W1, W2, W4 and W5 starting from November 1987 (Malmer and Grip, 1990 and *in press*). This paper mainly deals with results from control catchments W3 and W6 and to some extent with data from the other catchments during the reference period before treatments. W3 was the control catchment for the forest fire area and W6 for the lightly selectively logged forest. Strictly no treatments were applied to the control catchments.

Field sampling and laboratory handling

All streamwater samples have been grab sampled in pre washed polyethene bottles at the outflow jet of the flumes for streamflow measurements for the catchments. During the reference period baseflows from each catchment were sampled biweekly. Starting September 1987 baseflows were sampled 3 times per week until January 1989 when intensity was lowered to weekly sampling. Systematic flow proportional sampling was not practiced throughout this study, but regular stormflow sampling for dissolved nutrient analysis started in

November 1987. During 9 months of treatment and plantation establishment stormflow were sampled at each stream at 33 events and in the following monitoring until December 1990 at another 33 events. Furthermore at 13 of the stormflow events between January until October 1988 several samples per event were taken to cover different stages of the hydrograph.

During the whole 5 year period, biweekly baseflow samples and some stormflow samples were frozen within a few hours of sampling and later flown to Sweden for analysis at the Department of Forest Ecology (formerly Departement of Forest Site Research) at the Faculty of Forestry of the Swedish University of Agricultural Science in Umeå, Sweden. In September 1987 analysis of major plant nutrients started at the Sabah Forest Industries' field laboratory in Mendolong only 3 km from the research area, making analysis possible on the same day or the day after sampling. Out of a total of 2365 streamwater samples for this study, 837 has been analyzed in Sweden and 1971 in the Mendolong field laboratory, which means that 440 samples has been analyzed at both locations as a quality check.

Occasional sampling of rain for chemical analysis were done during the whole study period. In total 21 samples were collected during different parts of the year. The sampling was done in the Mendolong camp 500 m.a.s.l. and 3 km from the research area. A pre washed polyethene collector was placed at 1.5 m height, well away from houses and vegetation, at the start of the rain. The sample was frozen directly after collection as the collector was filled. All rain chemistry samples were analyzed in Sweden.

Methods for measurements of waterstage in the catchment streams and the computation of streamflow and runoff has been reported by Malmer (1992).

Chemical analysis

In Sweden samples were analyzed as follows: major cations (K, Na, Ca, Mg, B, Al, Fe, Mn, Zn, Cu and Mo), Si and total S were analyzed by inductive-coupled plasma atomic emission spectrometry (ICP-AES, Perkin Elmer, Plasma II Emission Spectrometer). Samples for ICP-AES were pre treated with nitric acid. Potassium in samples after October 1987 has been analyzed by atomic absorption. Anions (SO_4-S, NO_3-N, Cl, Br, F and NO_2-N) were analyzed by high performance liquid chromatography (HPLC, Dionex 4000i). NH_4-N and total N were analyzed spectrophotometrically according to Morris and Riley (1963). PO_4-P and total P were analyzed by flow injection spectrophotometry according to Murphy and Riley (1962). Total N was oxidized to NO_3 according to Koroleff (1969) and total P formed to PO_4 according to Koroleff (1976). Organic and inorganic C were analyzed using carbon analyzer (Carlo Erba 400/P), shifting to ICP-AES according to Emteryd et al. (1991) for organic C after April 1989.

In Mendolong NH_4-N, NO_3-N, PO_4-P and total P were analyzed as in Sweden, but using manual spectrophotometry and 50 mm cuvettes for lowered detection limit. At the field laboratory K, Ca and Na were analyzed by ion specific electrodes (Orion) (Emteryd, 1989). Colour (mg Pt/l) was analyzed in Mendolong spectrophotometrically (Anon., 1971) starting from January 1988.

Electrical conductivity (20° C) and pH were analyzed at the field laboratory for all samples after November 1986. Before that samples were analyzed in Sweden. Colour of water samples was subtracted for at all spectrophotometric analysis both in Sweden and in the field laboratory.

Data analysis

The dataset (November 1985 - December 1990) used for the analysis presented here emanates for the reference period from data from analysis in Sweden. So does also all data during and after treatment (starting from November 1985) except for PO_4-P, total P, NH_4-N, NO_3-N, K, Ca, Na and colour where data from Mendolong were used. Data from the two labs coincided well for samples analyzed at both labs. For the regularly very low concentrations of PO_4-P, total P, NH_4-N and NO_3-N regression between the two labs were poorer due to higher detection limit in Sweden (10 ppb compared to 5 ppb in Mendolong) and transformations from inorganic to organic phases during transport and storage. Data on electrical conductivity and pH are from analysis in Sweden before November 1986 and from Mendolong after that.

Data on organic- N and S were calculated by subtracting the concentration of inorganic phases analyzed from total- N and S respectively. Concentrations of HCO_3 used for correlation analysis were calculated from temperature and pH of a streamwater in equilibrium with the CO_2 concentration of the atmosphere. This was justified by the turbulent streamwater and from a few samples analyzed for inorganic C. Also a $CaCO_3$ content of the bedrock lower than 0.02 % (Malmer, unpublished) speaks against weathering as a important bicarbonate source. The anion charge deficit was calculated as the difference between the cations (H, K, Ca, Mg, Na and NH_4) and the anions (Cl, SO_4, NO_3 and HCO_3) on equivalent basis. This anion charge deficit may be accounted for by organic anions, that were not analyzed.

Autocorrelations between concentrations sampled three times per week were not significant. Regression analysis between concentrations and streamflow at sampling was made. These regressions were seldom significant and if so had low strength, except for some elements during and after treatments. Therefore regression analysis with dummy technique (Malmer, 1992, after Gujarati, 1978) could not be used to detect differences in concentrations between streams. Instead differences between mean concentrations between different streams and at the same stream between baseflow and stormflow during 5 periods were tested using Tukey-Kramer multiple test (Kramer, 1956) following tests of homogeneity in variance.

The 5 periods used for the evaluation of the data here are the ones used by Malmer (1990 and 1992) with slight modifications, namely:

1/ reference period December 1985 - November 1987
2/ clear-felling and extraction period November 1987 - February 1988
3/ burning and plantation period March 1988 - July 1988
4/ first hydrologic year after treatments August 1988 - July 1989
5/ second hydrologic year and further August 1989 - December 1990

The distribution of mean specific discharges at sampling of baseflow and stormflow during different periods was compared with the actual streamwater discharge dynamics for the streams during 1985 - 1990 presented by Malmer (1992). Sampled mean baseflow discharge was found to be 0.5 to 1 time higher than actual mean baseflow discharge. The reason for this was mainly that all baseflow samples were sampled in the early morning, a time of the day which had a higher mean baseflow than other times of the day. Mean sampled stormflows coincided better with mean peak discharge than mean stormflow discharge. The difference in sampled and actual mean baseflow discharge was not considered as a problem for the evaluation of baseflow mean concentration as the concentration was generally not a function of streamflow. Nighter was the fact that sampled stormflow coincided with peak stormflow, as long as one consider it as such and not as a mean for the whole stormflow hydrograph.

To get a measure of association between different elements, a correlation analysis (Wilkinson, 1990) on equivalent basis was preformed on a dataset from the two control streams for the whole five year period. Only sampling dates with complete chemical analysis were used. This gave sample sizes of about 80 and a dataset only containing baseflow samples. The correlation matrixes were visualized as graphical cluster descriptions, were correlations significant at $p <= 0.01$ were incorporated.

Reynolds et al. (1990) reviewed methods for computation of dissolved losses from upland streams. They particularly stressed the requirements of details on the variations in nutrient concentrations with stream discharge rates. In the present study strong relations between concentrations and streamflow were neither established for the control catchment streams during the whole period of monitoring nor for the other streams during the reference period. Therefore rating curves or regression equations could not be used for computation of element discharge. For these streams and periods significant differences between stormflow and baseflow sample concentrations were also rare (Appendix 1). To account for seasonal effects, and to facilitate for the computation and comparison of data from treated streams, period weighing was made by calculating the discharge of elements by multiplying monthly means of baseflow concentrations and streamflow. For those streams and time periods where stormflow concentration did differ significantly from baseflow concentration, also discharge weighing was made by multiplying monthly stormflow concentrations with peak streamflow (Malmer, 1992) and monthly baseflow concentration with total streamflow subtracted with peak streamflow. The risk for underestimation of nutrient losses by unrecorded deep leakage from small headwater catchments has been discussed by e.g. Bruijnzeel (1990). Malmer (1992) discussed the possibility of this problem to appear in this study. Catchment budget derived mean yearly evapotranspiration for this study (W6: 1540 mm) was within the range of a South-East Asian mean lowland rain forest evapotranspiration of about 1459 mm (n=6, Bruijnzeel, 1990). Low soil permeability (Andersson, 1990) and relatively shallow soil depth also speak against extensive leakage.

Mean values of element concentration in wet deposition has been compared with the relative concentration in sea water according to Eriksson (1960) to get an indication of terrestrial influence on the rain sampled. Chemical input by wet deposition was calculated for W3 and W6 by multiplying yearly mean areal rainfall with mean concentrations for the whole five year period. There was no weighing of mean values for different parts of the year. That was justified by the low influence of monsoonal rain at the trial site. The most intensive rainfall periods were recorded at the equinoxes, and the monsoonal influence was shown to decrease with a very short distance and with the elevation from the coast at Sipitang (Wong, 1989). Malmer (1992) also showed the driest periods in the research area to occur during the regional monsoon periods in December - February and June - August. The discharge of elements in the streamwater has been deducted from the wet deposition to approximate the catchment net loss (-) or gain (+).

RESULTS

Element concentrations in streamwater and wet deposition

Baseflow mean concentrations for control catchments W3 and W6 for the whole monitoring period 1985 - 1990 are tabulated in Table I. Baseflow and stormflow concentrations for all six streams and different time periods as well as statistical differences between streams and different flows are tabulated in Appendix 1. Concentrations of B, Cu, Mo, Br, F and NO_2-N in all streams during all periods were at or below detection limits for the methods of analysis. Detection limits for these elements were 0.001, 0.005, 0.04, 0.2, 0.5 and 0.05 mg l^{-1} respectively.

Table I. Baseflow mean concentration (mg l^{-1}) for analyzed elements in the two control streams W3 and W6 for the period November 1985 to December 1990 at Mendolong, Sabah, Malaysia. (pH in pH-units, Conductivity in uS cm^{-1}, and Colour in mg Pt l^{-1}).

stream	element concentration (mg l^{-1})									
	NO_3-N	NH_4-N	N-tot	PO_4-P	P-tot	K	Ca	Mg	Na	Si
W3	0.042	0.028	0.74	0.010	0.018	1.06	1.70	1.56	2.62	3.51
W6	0.014	0.014	0.99	0.004	0.009	0.31	0.22	0.24	0.58	0.48
	Fe	Al	Cl	SO_4-S	S-tot	C-org	C-inorg[1]	Zn	Mn	
W3	0.11	0.04	0.36	2.29	2.29	4.75	3.51	0.012	0.007	
W6	0.40	0.28	0.25	0.19	0.35	13.81	0.60	0.016	0.009	
	pH	Cond	Colour	HCO_3[2]						
W3	6.34	35.5	14.5	3.05						
W6	4.87	15.0	25.9	0.12						

1. Only 55 samples
2. Calculated from pH and CO_2 in equilibrium with the atmosphere

In spite of the low concentration levels there were differences in levels between the two control streams and also the other streams during the reference period for some elements. Largest differences were found between the control streams W3 and W6. The other streams always had intermediate levels of concentrations, pH and electrical conductivity during the reference period. Catchments W3 and W6 are dominated by different soil types as well as different vegetation, while the others have an areal mix of the two soil types (cf. above).

W3 had significantly ($p<=0.05$) higher baseflow mean concentrations during the reference period than W6 in K, Ca, Mg, Na, Si, SO_4-S, and total S. Also baseflow means of pH and electrical conductivity were higher at W3 than at W6 during period 1. For all of these elements there were significant differences between the group of catchments W1 - W3 and W4 - W6 during the reference period (Table II).

Practically all dissolved carbon appeared as organic C in all streams during the reference period. W3 later made an exception with only 28 % organic C as a mean for the whole five year period. Also nitrogen showed high organic dominance with around 10% inorganic N of total N, while sulphur had low organic phase compared to the inorganic (Appendix 1). The only exception was W6 which had 30% organic S. For the whole period of measurement the ratio of organic C to organic N was 7.1 to 14.4 for W3 and W6 respectively. C/N ratio for the relatively dry reference period was between 13.4 (W4) to 23.1 (W6).

Table II. Significant differences at p <=0.05 in baseflow mean concentration between catchments struck by forest fire 1982/83 (W1 - W3) and catchments selectively logged 1981 (W4 - W6) during the reference period nov 1985 - nov 1987. (> forest fire catchment concentration significant higher, - no significant difference.

Si	W4	W5	W6
W1	-	-	-
W2	>	>	>
W3	>	>	>

K	W4	W5	W6
W1	-	-	>
W2	-	-	
W3	-	>	>

Ca	W4	W5	W6
W1	>	>	>
W2	-	-	-
W3	>	>	>

Mg	W4	W5	W6
W1	>	>	>
W2	>	>	>
W3	>	>	>

Na	W4	W5	W6
W1	-	-	-
W2	-	-	>
W3	>	>	>

SO_4-S	W4	W5	W6
W1	-	>	>
W2	>	>	>
W3	>	>	>

Stot	W4	W5	W6
W1	>	>	>
W2	>	>	>
W3	>	>	>

On the other hand concentration of organic- N and C were the highest during shorter and more humid periods, as period 4 (1988/89) which also had high C/N ratios of 79.1 (W3) and 55.2 (W6).

Element concentrations in wet deposition are presented in Table 3. Given are also calculated relative proportion derived from sea water and the remaining concentration emanating from non marine sources. The non marine relative contribution was low, except for Ca, K, total N and organic C. Si, Fe, Mn, Cu, Mo and total P all were at or below analytical detection limits.

Table III. Mean dissolved chemical concentrations in µg l⁻¹ of wet deposition at Mendolong, Sabah, Malaysia and calculated marine and non marine contributions for some elements. (N = 21)

	element concentration (µg l^{-1})										
	Cl	SO_4-S	S-tot	Na	K	Ca	Mg	N-tot	Corg	Al	Zn
total in bulk precipitation	185	56	86	124	52	48	10	183	1577	33	15
maritime origin	185	26	26	103	4	4	12[1]				
non-maritime origin	0	30	60	21	48	44	0[1]				

1. Difference between concentration in bulk precipitation and maritime origin within detection limit for analysis.

Both element concentrations and total input in kg ha^{-1} calculated from the sampled rainfall were low compared with other studies (Zulkifli Yousop et al., 1989; Bruijnzeel, 1989). The precipitation in the area is dominated by convective rains, giving a low input of marine origin (Table III). The generally low concentrations also indicate the absence of anthropogenic pollution. The non marine input was dominated by organic C and total N, indicating some low organic particle input. The non detectable levels of Si and Fe indicate a lack of mineral particles in the input, despite that rain chemistry was sampled in the Mendolong camp with surrounding heavily trafficked logging roads and large plantation areas with disturbed soils. Inorganic fractions of nitrogen is not presented due to possible transformations to organic phase during transport and storage before analysis in Sweden.

Cluster analysis of baseflow concentrations

The clusters formed by the correlations between the different elements in baseflow between reference streams W3 and W6 were rather different (Figure 2). In W6 (Figure 2a), the main cluster was formed by the positive correlation between Mg, Si, anion charge deficit (cf. *data analysis* above), organic- S and N with Cl, SO_4 and organic C loosely associated. A weaker cluster was formed by organic C, NH_4 and Zn. Al and Fe formed a strong cluster with no significant correlation to the main cluster. Ca, Na, K, NO_3, PO_4 and streamflow had no significant ($p <= 0.01$) associations.

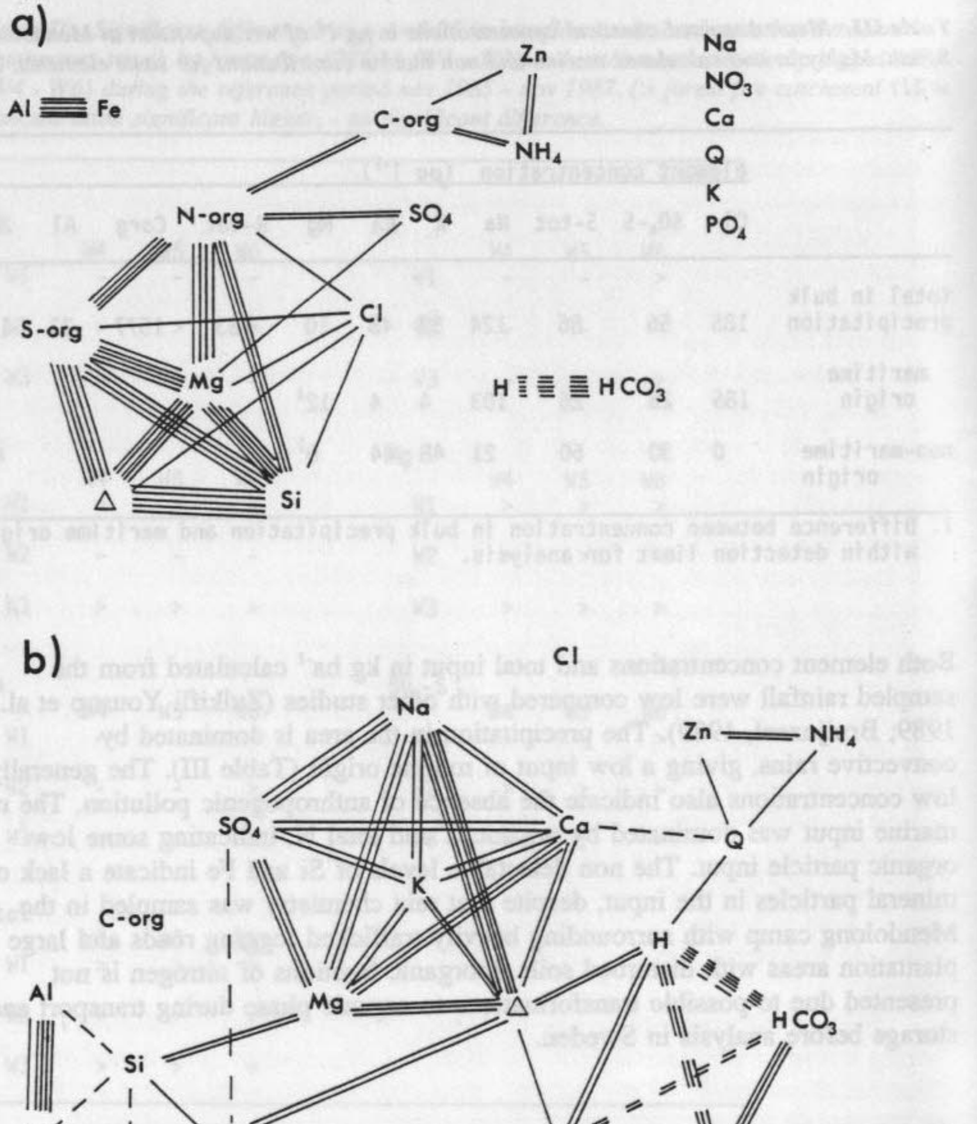

Figure 2. *Cluster description of correlations between concentrations of different chemical constituents, anion charge deficit (**Δ**) and streamflow (Q) for control catchment streams W6 (a) and W3 (b) for baseflow during the period November 1985 to December 1990. Only significant correlations (p<=0.01) are shown. 0.3<r<=0.4 /; 0.4<r<=0.5 //; 0.5<r<=0.6 ///; 0.6<r<=0.7 ////; 0.7<r /////. Negative correlations have broken lines.*

In W3 (Figure 2b), the main cluster was formed by Mg, anion charge deficit, Ca, Na, SO_4 and K. To this main cluster Si, organic S, PO_4, H^+ were positively correlated, while streamflow was negatively correlated. Si formed a secondary cluster with negative correlations to organic N and the strongly linked Al and Fe and positive correlation to organic S. To anion charge deficit in the main cluster, PO_4 and H^+ were positively correlated, and they in turn were negatively correlated to NO_3 and HCO_3. H^+ and Ca were linked by streamflow to a third peripherical cluster, containing NH_4 and Zn.

Catchment element discharge, input and net loss or gain in streamwater

Mean yearly dissolved element discharge (kg ha^{-1} y^{-1}) from the control catchment streams W3 and W6 are presented in Table IV. W6 had markedly higher discharge than W3 in organic- N, C, and S together with H^+, Al, Fe and Zn. W3 on the other hand had larger discharge of especially the cations K, Ca, Mg and Na as well as for Si, P, NO_3 and inorganic- C and S.

Chemical input through wet deposition for W3 and W6 are also tabulated in Table IV, together with approximated net loss or gain of respective element after deducting the dissolved discharge in streamwater from the wet deposition. Phosphorus was not detectable (<10 ug l^{-1}) in bulk precipitation input. The maximum input at detection level with 3400 mm annual rainfall is 0.03 kg ha^{-1} y^{-1}. Deducting that from streamwater discharge means a maximum net loss of 0.27 and 0.12 kg ha^{-1} y^{-1} of total P for W3 and W6 respectively. The same type of calculation were made for other elements not detectable in the wet deposition (Si, Fe, Mn and inorganic C).

The mean yearly dissolved net loss from the two catchments were higher from W3 with 0.25 t ha^{-1} y^{-1} compared to 0.16 t ha^{-1} y^{-1} for W6. Also particulate losses from W3 (0.48 t ha^{-1} y^{-1}) was higher than from W6 (0.22 t ha^{-1} y^{-1}) (Malmer, 1990 and this study). In Table V also data on stream dissolved macro nutrient net losses from other studies in similar environments on soils of low to moderate fertility (after Bruijnzeel, 1990) are compared with catchment net losses from this study. The levels of discharge from the Podsolic W6 was quite similar to the ones from the Spodosol site, but net losses/gains rather different, although at both sites on a low level. At W3 the levels were higher than means from other moderate to low fertility sites.

***Table IV**. Mean yearly dissolved stream discharge, catchment wet deposition input and catchment net loss or gain (kg ha^{-1} y^{-1}) from control catchments W3 and W6 at Mendolong, Sabah, Malaysia during a 56 month period. Total dissolved net losses from this study is tabulated together with total particulate losses (kg ha^{-1} y^{-1}). (nd = not detected, na = not analyzed)*

element	discharge W3	discharge W6	input W3	input W6	net loss(-) or gain(+) W3	net loss(-) or gain(+) W6	W3/W6
NO_3-N	0.38	0.10					
NH_4-N	0.29	0.22					
N-tot	2.60	4.47	5.9	6.4	+3.3	+1.9	1.7
PO_4-P	0.19	0.07					
P-tot	0.30	0.15	nd	nd	-0.27[1]	-0.12[1]	2.2
Na	30.7	7.1	4.0	4.3	-21.5	-1.4	15.4
K	12.4	3.2	1.7	1.8	-10.7	-1.5	7.1
Ca	19.7	3.1	1.5	1.7	-18.2	-1.6	11.4
Mg	17.3	3.2	0.3	0.4	-17.0	-2.9	5.9
C-org	85.9	193.8	50.7	55.0	-35.2	-138.8	0.2
C-inorg[2]	63.5	8.4	nd	nd	-61.3[1]	-4.9[1]	12.5
Si	48.6	6.2	nd	nd	-48.3[1]	-5.8[1]	8.3
Fe	1.2	4.7	nd	nd	-1.0[1]	-4.5[1]	0.2
Al	0.4	3.2	1.1	1.2	+0.7	-2.0	-0.4
Zn	0.2	0.3	0.5	0.5	+0.3	+0.2	1.5
Mn	0.1	0.1	nd	nd	-0.1[1]	-0.1[1]	1.0
SO_4-S	24.0	2.0	1.8	2.0	-22.0	+/-0	
S-tot	24.3	4.4	2.8	3.0	-21.5	-1.4	15.4
Cl	4.6	3.0	6.0	6.5	+1.4	+3.5	0.4
H[3]	0.01	0.3	na	na			
Total loss[4]					251.4	159.4	1.6
Particulate losses[5]					481	221	2.2

1. Net loss or gain calculated by subtracting wet deposition at detection level concentration with element discharge.
2. Estimated from the relation between period mean concentrations of organic- and inorganic C times loss of organic C.
3. Estimated from period mean pH and total period runoff.
4. Total from this table, except H and inorganic fractions of N, P and S.
5. (Malmer, 1990, and unpublished data for 1989/90).

DISCUSSION

Relations of element concentration in streamwater

The Gleyic Podsol in W6 had a much lower field saturated hydraulic conductivity in the top soil layer than the Orthic Acrisol in W3 (0.11 mm h^{-1} compared with 0.68 mm h^{-1} at 20 cm depth; Andersson, 1990). The component of infiltration excess and saturated overland flow should therefore be larger in the low laying parts of W6 than in the same parts in W3, which could also be concluded from Malmer (1992). This indicates that the mean transit time for water is longer in W3 than in W6, which also affects the differences in streamwater baseflow mean concentrations (Table I). The different hydrology also affects the conditions for mineralization of organic material, as larger areas of top soil could be expected to be saturated for longer periods in W6. This may be the reason why dissolved organic material is discharged together with Mg, Si, Cl and SO_4 in the main cluster of W6. A positive relation between the ratio of dissolved organic carbon and organic nitrogen in streamwater as a function of areal % podsol at different catchments before treatments was found (Figure 3).

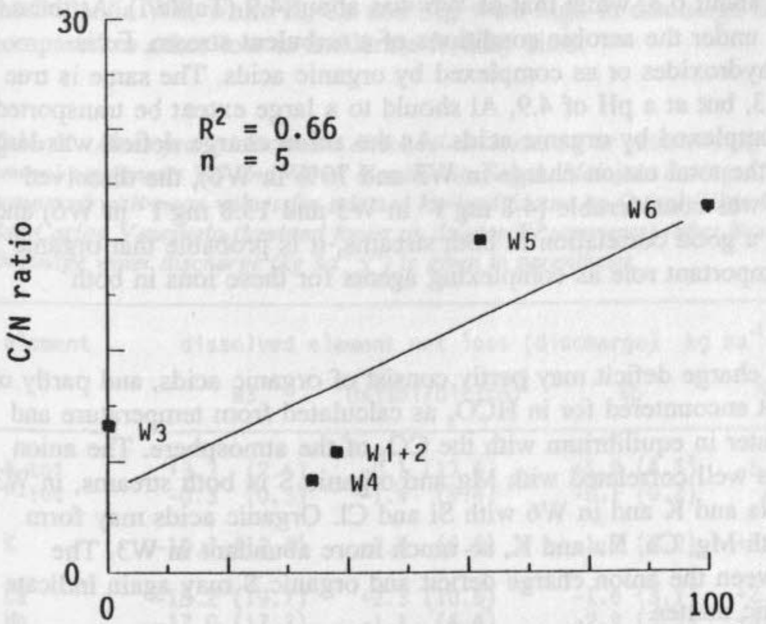

***Figure 3.** The dependence of the dissolved organic C/N ratio in streamwater on the relative areal coverage of Gleyic Podsol (%) in the catchments W1 - W6 at Mendolong, Sabah, Malaysia during the reference period November 1985 to November 1987.*

The C/N ratio of humus tends to move towards that of microbial cells during its mineralization. This ratio is about 10, but values between 5 to 15 are not uncommon (Alexander, 1977). As long term means during pretreatment the dissolved organic matter in W3 seemed to have reached its final stage of decomposition, while that of W6 was less decomposed.

The main cluster of baseflow concentrations in W3 was more geochemical than in W6, as it contained the weathering products Ca, Mg, Na, K and SO_4. Si was also associated to this group, which was negatively correlated to streamflow, as is common in boreal forest ecosystems (e.g. Grip, 1982). As in boreal forests, H^+ increased with streamflow. The organic compounds were only weakly associated to the main cluster and often negatively. (organic C was negatively correlated to Mg at $p <= 0.05$).

In both streams Al and Fe were strongly correlated, but only weakly and negatively to the main clusters (in W6, Fe was negatively correlated at $p <= 0.05$ to SO_4 and organic N). Weathering is the source of both these elements, but from the clustering it can be concluded that they have a different transport mechanism compared to the other ions of the same source. The mean pH of W3 was about 6.3, while that of W6 was about 4.9 (Table I). At these pH-values, and under the aerobic conditions of a turbulent stream, Fe is transported as hydroxides or as complexed by organic acids. The same is true for Al at pH 6.3, but at a pH of 4.9, Al should to a large extent be transported as Al^{3+} or as complexed by organic acids. As the anion charge deficit was large (about 40% of the total cation charge in W3 and 70% in W6), the dissolved organic carbon was considerable (4.8 mg l^{-1} in W3 and 13.8 mg l^{-1} in W6) and Al-Fe has such a good correlation in both streams, it is probable that organic acids play an important role as complexing agents for these ions in both streams.

The anion charge deficit may partly consist of organic acids, and partly of bicarbonate, not encountered for in HCO_3 as calculated from temperature and pH of streamwater in equilibrium with the CO_2 of the atmosphere. The anion charge deficit is well correlated with Mg and organic S in both streams, in W3 also with Ca, Na and K and in W6 with Si and Cl. Organic acids may form associations with Mg, Ca, Na and K, so much more abundant in W3. The correlation between the anion charge deficit and organic S may again indicate dissolved organic matter.

The ammonium ion was correlated to stream baseflow (in W6 significantly to streamflow for $p <= 0.05$). The ammonium source is dcomposing organic material at or close to the soil surface and it is only lost in the form of ammonium if it is transported to the stream by superficial water with short transit time. Thus it is natural that it is positively correlated to streamflow.

Dissolved element discharge and catchment loss or gain

As for major cations also element discharge in streamwater was rather similar in this study compared with other studies (Table V). However, discharge of N and P in streamwater was much lower than from other studies on comparable soils. Especially the low concentrations of P and inorganic N in natural waters rise a demand for low detection limits and high standards of transport, storage and analysis to avoid contamination and transformations (e.g. Jordan, 1982; Lewis, 1986). Very variable and sometimes comparably high losses of P, described in earlier studies, might not only depend on differences in soils and vegetation, but also on differences in analysis. Discharge of K, Ca and Mg from the Spodosol site in Venezuela (Table IV) was quite similar to that of the Podsolic W6, while K, Ca and Mg were high in discharge from W3 compared to other low to moderate fertility sites.

***Table V.** Mean yearly catchment dissolved net losses (-) or gains (+) (kg ha^{-1} y^{-1}) from control catchments W3 and W6 at Mendolong, Sabah, Malaysia during a 56 month period, compared with mean values for selected lowland forests on Oxisols/Ultisols and data from San Carlos, Venezuela (lowland forest on Spodosol/Psamments) (after Bruijnzeel, 1990). Drainage water discharge (kg ha^{-1} y^{-1}) is given in parenthesis.*

element	dissolved element net loss (discharge) kg ha^{-1} y^{-1}			
	W3	Oxisol/Ultisol[1]	W6	Spodosol[2]
N-tot	+3.3 (2.6)	-0.1 (13.4)	+1.9 (4.5)	-5.9 (8.2)
P-tot	-0.3 (0.3)	-1.4 (5.4)	-0.1 (0.2)	- -
K	-10.7 (12.4)	+2.0 (8.6)	-1.5 (3.2)	+3.4 (3.5)
Ca	-18.2 (19.7)	+2.3 (10.5)	-1.6 (3.1)	+2.4 (2.8)
Mg	-17.0 (17.3)	-1.1 (4.4)	-2.9 (3.2)	+0.1 (0.6)

1. N=; 11 (K, Ca and Mg), 8 (P) and 6 (N). Calculated after Bruijnzeel (1990, Table III). There presented "alternate computations" were used instead of original data. Data on losses of P and N were excluded in cases were only fractions of P-tot or N-tot were presented.
2. "Alternate computation" by Bruijnzeel, 1990 (after Herrera, 1979 and Buschbacher, 1984).

After subtracting streamwater dissolved discharge from input in wet deposition, the losses (-) or gains (+) of different elements for the control catchments W3 and W6 were very different compared with each other (Table IV). Bruijnzeel (1990) showed ranges of K, Ca and Mg discharge from some 20 selected (sub) tropical ecosystems to relate to soil fertility and annual runoff, with the most effective nutrient conservation on low fertility and with low humidity. Higher weathering in deeper horizons, not reach for plant roots, can also be expected to be larger from deep, strongly weathered soils (cf. Burnham, 1989). Dissolved discharge from both catchments in this study fall well within observed ranges for comparable soil fertility groups in that comparison. Adding the comparison in Table V it can be seen that the net losses from W3 in K, Ca and Mg were in the upper part of the range compared to a mean on comparable soils.

The fact that Cl seems to be accumulating in small amounts in both systems is hard to explain from common knowledge. Chloride budgets are often used to detect and determine deep leakage from catchments (e.g. Eriksson, 1985). In this study extensive deep leakage is not considered likely (c.f. above and Malmer, 1992). Another reason could be overestimation of the input of Cl. A lower concentration of elements from the marine source during short periods might have been overlooked in this study due to a low number of rain chemistry samples. However, the samples covered different seasons, and the concentrations in this study are already very low by comparison. Instead, extensive losses of Cl from the other catchments in this study after clear-felling (Malmer and Grip, *in press*) indicated that the accumulation showed here was real.

Also nitrogen was accumulating in both control catchments. The nitrogen accumulation in W3 is almost double that of W6. That might be attributed to the probably higher biomass accumulation in the secondary succession of vegetation of W3 compared to the dipterocarp forest of W6. It should be noted that nighter biological nitrogen fixation, nor denitrification is considered here. Podsols in the tropics have been shown to have constraints on decomposition compared to more fertile soils (e.g. Vitousek and Sanford, 1986). As discussed above the conditions for top soil mineralization in the Podsol may be worse than in the Acrisol in this study. Therefore the uptake by the vegetation on the Podsol could also be expected to be slower.

Adapting a maximum P input at detection limit concentration (c.f. above) results in a small net loss of P from both W3 and W6. The P discharge from W3 was, even though small, on a markedly higher level than from W6 and the calculated net loss from W3 was 4 times larger than from W6. Most other studies show accumulation of P relating to the low mobility of the element (Bruijnzeel, 1990). However the low levels for analysis may bias many results (c.f. above), and the net loss here might be biased by the insecure figure on input. The difference in larger discharge from W3 compared to W6 is more secure and based on many samples during a long time period and low level detection analysis.

Causes for differences in nutrient losses or gains

Apart from the differences in streamwater content of organic C discussed above, the most striking difference in losses between the control catchments concerned total S, Si and the cations Ca, Na, K and Mg. W6 showed very small losses while the losses from W3 were 6 to 15 times larger (Table IV). These differences could be caused by differences in nutrient conserving mechanisms between the forest and the secondary vegetation, by the difference in soil type and by local differences in bedrock. A higher element loss from W3 due to the forest fire 2.5 years before the start of streamwater sampling is not considered a main cause here. A high loss only directly after the fire could be assumed, as the significant effects on streamwater concentrations of the treatment induced by the fires in the treatments in this study had practically sized 1 year after burning (Appendix 1). Rather the gradient with intermediate concentrations from W1+2 with a mix of soil types and a harder burn of the forest fire speaks for the soil type being decisive for differences in losses.

If, as assumed above, the accumulation of nitrogen is higher in W3, it could be assumed that the secondary vegetation under succession also accumulates other elements to a similar higher rate compared to the forest of W6. This together with the higher losses of cations from W3 indicate a much higher total input from weathering in W3. One important variable governing the difference in weathering between the Orthic Acrisol and the Gleyic Podsol is the relative area of particle surfaces, which would be expected to be much smaller in the Podsol due to textural differences (Malmer and Grip, 1990 and above).

The fact that the dark shales seem to be more common under W3 (cf. above) also makes the nutrient content of the bedrock to a probable cause to higher weathering input to W3. A test of hydrogenfluoride digestion of some bedrock samples (Malmer, unpublished) revealed ratios for element contents between the shale and the sandstone like 1.7 (K), 4.2 (Na), 3.1 (Ca), 10.8 (Mg) and 45.3 (S). This explains the more than tenfold loss of S from W3. As the ratios of cations between the two rock types is not as large as the differences in losses (except for Mg), it can be concluded that both bedrock nutrient content and weathering rate in the different soil types make up the differences in weathering input to W3 and W6.

CONCLUSIONS

Local differences in nutrient content of the bedrock together with different soil type weathering rate was found decisive for weathering input and thus for streamwater concentration and net losses of weathering products, like Ca, Mg, Na, K, Si, Fe, Al, SO_4, HCO_3 and also for pH. By difference in hydraulic conductivity the Gleyic Podsol resulted in more superficial water flow pathways than the Orthic Acrisol. Therefore, the stream from the Podsol had higher content of dissolved organic material. As seen from the C/N ratio this organic material was also less decomposed than in the stream from the Acrisol. Nitrogen was found to accumulate in both catchments. An almost double rate of nitrogen accumulation in W3 was probably connected with the accumulation of biomass in the secondary succession of vegetation after the forest fire 3 - 8 years earlier.

A low net loss of phosphorous was found from both systems during the study period. This net loss was 4 times higher from the secondary vegetation than from the rainforest.

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Appendix 1.

Baseflow mean concentrations (mg l^{-1}) for reference period (1) and periods during(2,3) and after (4,5) treatment. Any period mean stormflow concentration significantly different at p <=0.05 from baseflow concentrations are tabulated i a second row for the respective period. In those cases where concentration was significantly correlated at p <=0.01 to streamflow by linear regression (R^2 >0.3) this is marked by *. (<,> significant difference in concentration between treated and reference stream, @ concentration below detection limit)

element/ /period	catchment w1	w3	w2	w4	w6	w5
NO_3-N 1	0.029	0.063	0.048	0.030	0.071	0.041
2	0.019	0.043	0.057*	0.072*	0.011	0.106
	0.221		0.291	0.341		
3	0.048*	0.034	0.204	0.275 >	0.008 <	0.151
	0.716		0.634	0.531		0.454
4	0.006	0.028	0.039	0.146 >	0.008 <	0.232
5	0.008	0.027	0.007	0.062	0.011	0.125
NH_4-N 1	0.008@	0.005@	0.009@	0.009@	0.010	0.011
2	0.038	0.021	0.022	0.067	0.017 <	0.252
				0.243		1.297
3	0.023	0.015	0.017*	0.048	0.010 <	0.131*
	0.785		0.662	0.638		0.592
4	0.040	0.017	0.017	0.036	0.016	0.031
5	0.022	0.032	0.028	0.056	0.028	0.030
Ntot 1	0.324	0.252	0.258	0.364	0.427	0.368
2	0.281	0.201	0.348*	0.392	0.331	0.526
3	0.402	0.314	0.568	0.771*	0.673 <	1.112
	2.239	1.888	1.995	2.200		
4	0.356	0.151	0.240	0.878 >	0.381 <	0.812
5	0.367	0.250	0.271	0.460	0.368	0.590

element/ /period	catchment W1	W3	W2	W4	W6	W5
PO_4-P						
1	0.006@	0.010	0.008@	0.007@	0.003@	0.006@
2	0.005	0.007	0.006	0.007	0.002@ <	0.022
3	0.007* 0.284	0.005	0.009* 0.352	0.004@ 0.134	0.004@	0.007* 0.123
4	0.006	0.012	0.009	0.006	0.004@	0.007
5	0.009	0.019	0.016	0.012	0.007	0.012
Ptot						
1	0.013	0.016	0.009@	0.012	0.004@	0.007@
2	0.011	0.009	0.015	0.013	0.006	0.032
3	0.019* 0.305	0.013	0.018* 0.332	0.011 0.116	0.007	0.015* 0.156
4	0.015	0.022	0.019	0.016	0.010	0.018
5	0.017	0.027	0.022	0.022	0.013	0.020
K						
1	0.92	1.09	0.91	0.48	0.36	0.45
2	2.00 7.48	0.97	1.41	2.29* > 6.01	0.22 <	3.86 6.51
3	3.54* > 40.22	1.17	3.55 45.64	2.99* 9.22	0.32 <	5.36* 13.24
4	1.78	0.98	1.59	2.45	0.27	2.92
5	0.86*	1.02*	0.87	1.43	0.22	1.32
Ca						
1	1.89	2.17	1.26	0.31	0.26	0.31
2	1.43	1.72 >	1.05	0.41	0.12	0.78
3	2.23 5.85	1.77 >	1.33 5.88	0.56 3.12	0.27 <	0.90
4	1.82	1.56	1.08	0.53	0.26	0.71
5	1.36	1.57	1.04	0.46	0.14	0.56

element/ /period	catchment W1		W3		W2	W4		W6		W5
Mg										
1	1.07	<	1.76	>	1.14	0.36		0.16		0.29
2	0.76	<	1.52	>	0.84	0.49		0.20		0.68
3	1.11		1.53	>	0.72	0.84		0.34	<	1.03
4	0.85		1.25		0.76	0.52		0.48		0.54
5	0.72	<	1.48		0.89	0.45		0.14		0.44
Na										
1	1.15	<	3.24	>	1.67	1.16		0.70		1.18
2	0.99	<	3.00* 1.66		1.54	1.23	>	0.71	<	1.16*
3	0.72		2.40 1.36	>	1.15	0.86		0.44		0.93
4	0.60	<	2.03		0.81	0.72		0.39		0.70
5	1.04	<	3.68	>	1.77	1.29		0.84		1.54
Si										
1	0.96	<	3.10	>	1.93	0.54		0.38		0.49
2	1.30	<	2.73 0.23		1.91	1.34		1.19		1.40
3	1.27	<	3.85	>	1.76	1.47		0.52	<	2.30 1.23
4	0.73	<	3.82	>	1.79	0.51		0.60		0.61
5	0.88	<	4.02	>	2.34	0.72		0.33		0.74
Fe										
1	0.32		0.10		0.14	0.42		0.40		0.37
2	1.59 0.30	>	0.32		0.44	1.23		0.74	<	3.95 0.70
3	0.36		0.07*		0.19	0.18		0.17		0.30
4	0.69		0.10		0.17	0.54		0.41		0.77
5	0.34		0.05		0.12	0.40		0.36		0.49

element/ /period	W1	W3	W2	W4	W6	W5
Al						
1	0.06	0.03	0.04	0.08	0.28	0.15
2	0.19* 0.41	0.11	0.11	0.40	0.43 <	1.75
3	0.11	0.04 0.15	0.15	0.10	0.26	0.25
4	0.12	0.03 0.06	0.07	0.12	0.24	0.24
5	0.07	0.02@	0.03	0.09	0.23	0.12
Cl						
1	0.31	0.33	0.34	0.24	0.23	0.24
2	0.42 2.07	0.24	0.38	0.97 > 1.77	0.19@ <	1.08
3	0.98 > 3.38	0.22	0.56	1.34* > 2.46	0.33 <	1.66
4	0.11@	0.44	0.20	0.77	0.24	0.72
5	0.13@	0.42	0.14@	0.31	0.32	0.44
SO_4-S						
1	1.65	2.65	2.12	0.74	0.18	0.64
2	1.03 <	2.17	1.82	0.64	0.15	0.38
3	1.03 < 5.59	2.26	1.72	0.71	0.30	0.75
4	0.75	1.92	1.29	0.76* 1.04	0.20	0.52
5	1.16	2.89	1.42	1.00	0.22	0.69
Stot						
1	1.78 <	2.65	2.12	0.76	0.26	0.64
2	1.08	2.37	1.82	0.86	0.31	0.71
3	1.03 5.96	2.26	1.72	1.46	0.51	1.47 2.66
4	0.75	1.92	1.29	0.76	0.57	0.65
5	1.16	2.89	1.42	1.00	0.22	0.70

element/ /period	catchment W1	W3	W2	W4	W6	W5
Corg						
1	4.80	1.37	2.73	3.40	7.95	5.23
2	3.60	1.94	3.01 23.21	3.83	9.73	9.90
3	20.80 >	10.34	10.68	12.72	22.38	17.99
4	13.31	9.92	12.10	12.72	17.79	14.79
5	10.06	3.02	4.90	8.57	15.12	11.99
Zn						
1	0.007	0.005	0.009	0.009	0.009	0.018
2	0.005	0.012	0.035 0.200	0.014	0.007	0.043
3	0.025	0.019	0.026	0.027	0.032	0.036
4	0.014	0.006	0.011	0.065	0.012	0.013
5	0.044	0.028	0.038	0.025	0.033	0.032
Mn						
1	0.036 >	0.006	0.018	0.016	0.009	0.012
2	0.030* >	0.006	0.012	0.029	0.015 <	0.059
3	0.034 0.104	0.017	0.010	0.020*	0.013	0.012 0.042
4	0.031	0.010	0.008	0.021	0.007	0.011
5	0.031	0.005	0.011	0.019	0.008	0.009
pH						
1	6.04 <	6.75 >	6.30	5.96 >	4.76 <	5.62
2	5.79 <	6.42 >	6.00	5.76 >	4.75 <	5.51
3	5.88 6.36 <	6.25 6.63	5.99 6.54	5.86 >	4.96 <	5.81
4	5.86	6.20	5.99 5.78	5.82 >	4.92 <	5.75
5	5.61 <	6.06	5.90	5.75 >	4.97 <	5.69

element/ period	catchment W1	W3	W2	W4	W6	W5
Cond ($\mu S\ cm^{-1}$)						
1	22.1	< 37.5 >	24.4	12.1	14.8	12.4
2	25.2	< 37.5 >	25.5	19.6	14.2 <	23.3
3	33.3 106.5	37.7	32.2 196.3	23.0 76.4	> 14.0 <	30.1
4	24.2	< 33.0 >	22.9	21.5	> 14.7 <	22.3
5	22.3	< 40.4 >	25.2	18.0	15.7	18.1
Colour (mg Pt/l)						
1	-	-	-	-	-	-
2	22.61	7.85 59.18	8.49	24.63 83.00	22.67	< 126.51 347.75
3	23.72 298.67	7.62 150.60	8.17	19.20 217.22	22.00	53.25 455.83
4	25.10	8.36	15.84	21.49	26.37	39.58 107.16
5	17.54	9.58	11.57	23.69	24.37	31.49

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CONVERTING TROPICAL RAINFOREST TO FOREST PLANTATION IN SABAH, MALAYSIA. II. EFFECTS ON NUTRIENT DYNAMICS AND NET LOSSES IN STREAMWATER.

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ABSTRACT

Streamwater chemistry was monitored during five years in six streams in a paired catchment experiment in Mendolong, Sabah, Malaysia, comparing effects of different ways to establish forest plantations with *Acacia mangium*. At the start of the monitoring in 1985 three catchments were covered with selectively logged rainforest (W4 - W6) and three with secondary vegetation after forest fire (W1 - W3). Treatments were:

1. slashing of secondary vegetation, burning and planting (W1 and W2)
2. clear-felling, crawler tractor extraction, burning and planting (W5)
3. clear-felling, manual extraction, no burning and planting (W4)

W3 and W6 were monitored as control catchments. Calibration monitoring in all streams was 1 [illegible] years, treatments until new vegetative cover lasted for 9 months. Th[illegible] monitoring for another 2.5 years. The soil type of the catchments was [illegible] and Gleyic Podzol in W6 and a mix of both soil types in the other ca[illegible]

Effect of treatments on streamwater chemistry was clear in both [illegible] Concentrations of major plant nutrients like N, P and K became positively correlated to streamflow during treatments. Response of leaching from slash at clear-felling was fast and larger from the clear-felling treatment (W4 and W5) than the slashed secondary vegetation (W1 and W2). The response of burning was more intense. Stormflow period mean concentration was approximately 10-fold for N and K and 50 - 100 fold for P after burning compared with baseflow mean concentration in the same period. Significant differences in baseflow concentrations in treated streams lasted generally 1 year for most elements, but elevated concentrations were still detectable after 5 years. The first large pulse of leaching was connected to the mineralization after felling and particularly burning, and the longer lasting elevated concentrations in baseflow with loss of weathering products.

Amounts of nutrients lost, calculated by regression analysis as effect of treatment compared to control, was concluded to be higher with the degree of vegetation killed and with increased soil disturbance. Consequently normal practice with crawler tractor extraction and burning before planting created the largest leaching losses. Total calculated effect of losses in total N, P and K was for W1+2: 0.5, 1.6, 83.9; W4: 0.8, 0.2, 103.6; W5: 1.3, 1.5, 169.4 kg ha^{-1} for the period of 33 months during and after treatment. For the normal practice with crawler traction and burning before planting (W5) the treatment induced loss of K was equivalent to 86% of the content of easily decomposed parts of the biomass (leaves, twigs, fine roots and ground vegetation) of the old forest. Exhaustion effects of increased leaching after repeated burning (forest fire and pre planting fire) was observed for several elements indicating possible deficiencies.

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II. EFFECTS ON NUTRIENT DYNAMICS AND NET LOSSES IN STREAMWATER.

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Streamwater chemistry was monitored during five years in six streams in a paired catchment experiment in Mendolong, Sabah, Malaysia, comparing effects of different ways to establish forest plantations with *Acacia mangium*. At the start of the monitoring in 1985 three catchments were covered with selectively logged rainforest (W4 - W6) and three with secondary vegetation after forest fire (W1 - W3). Treatments were:

1. clearing of secondary vegetation, burning and planting (W1 and W2)
2. clear-felling, crawler tractor extraction, burning and planting (W5)
3. clear-felling, manual extraction, no burning and planting (W4)

W3 and W6 were monitored as control catchments. Reference monitoring at all streams was 2 years, treatments until new vegetative cover lasted for 9 months. This paper covers monitoring for another 2.5 years. The soil type of the catchments were Orthic Acrisol in W3 and Gleyic Podsol in W6 and a mix of both soil types in the other catchments.

Effect of treatments on streamwater chemistry was clear at both base- and stormflows. Concentrations of major plant nutrients like N, P and K became positively correlated to streamflow during treatments. Response of leaching from slash at clear-felling was fast and larger from the clear-felling residues (W4 and W5) than the cleared secondary vegetation (W1 and W2). The response of burning was more intense. Stormflow period mean concentration was approximately 10-fold for N and K and 10 - 100 fold for P after burning compared with baseflow mean concentration at the same period. Significant differences in baseflow concentrations in treated streams lasted generally 1 year for most elements, but elevated concentrations were still detectable after 3 years. The first large pulse of leaching was connected to the mineralization after felling and particularly burning, and the longer lasting elevated concentrations in baseflow with loss of weathering products.

Amounts of nutrients lost, calculated by regression analysis as effect of treatment compared to control, was concluded to be higher with the degree of vegetation killed and with increased soil disturbance. Consequently normal practice with crawler tractor extraction and burning before planting created the largest leaching losses. Total calculated effect of losses in total N, P and K was for W1+2: 0.5, 1.8, 83.9; W4: 0.8, 0.8, 105.6; W5: 1.3, 1.3, 189.4 kg ha^{-1} for the period of 33 months during and after treatment. For the normal practice with crawler tractors and burning before planting (W5) the treatment induced loss of K was equivalent to 86% of the content of easily decomposed parts of the biomass (leaves, twigs, fine roots and ground vegetation) of the old forest. Exhaustion effects of lowered leaching after repeated burning (forest fire and pre planting fire) was observed for several elements indicating possible deficiencies.

INTRODUCTION

The last decades has seen an increased trend towards plantation forestry and extractive tree crops in the tropics (Whitmore, 1984). The increment of areas of forest plantations can also be expected to continue ahead in the future (Evans, 1982; Whitmore, 1990). The evaluation of sustainability of productivity in terms of nutrients for a tree plantation is complex and especially data on hydrological nutrient losses and rates of chemical weathering are scarce from tropical forests and plantation establishment (Bruijnzeel, 1990; Anderson and Spencer, 1991).

The present paired catchment study, started in 1985, is monitoring the hydrological, hydrochemical, soil and biomass changes before, during and after different ways of converting selectively logged tropical rainforest and forest struck by forest fire to forest plantation. Grip et al. (*in press*) described the stream chemistry and catchment nutrient discharge from the control catchments. Areas on Podsol had tight nutrient circulation with a small accumulation of nitrogen, while areas on Acrisol and richer shales had markedly larger losses of major cations and sulphur due to larger input from weathering. A larger accumulation of nitrogen in the secondary vegetation was attributed to the regrowth after the forest fire.

This paper presents data on changes in dynamics and net losses of dissolved nutrients in streamwater leaving the treated catchments.

RESEARCH AREA

The Mendolong research area including the six catchments used for this impact study is situated at 650 - 750 m.a.s.l. on the foothills of Mount Lumako in the Crocker Range 35 km south east of the coast at Sipitang (115.5° E, 5.0° N) in Sabah, Malaysia. The natural forest of the research area is a lowland hill dipterocarp forest (Whitmore, 1984, after Symington, 1943), although the altitude is just below the transition to lower montane forest. The size of the catchments are between 3.4 and 18.2 ha, and the treatments being compared cover 100% of the respective basins.

Slopes in the area are mostly moderate, ranging up to 27%, but steeper slopes of up to 57% also occur. The research area includes two different soil types, Orthic Acrisol and Gleyic Podsol. Control catchment W3 is wholly dominated by the Orthic Acrisol, while W6 is mainly consisting of the Gleyic Podsol. The other catchments have an areal mix of the two soil types. More details on soils and bedrock were given by Malmer and Grip (1990) and Grip et al. (*in press*).

The rainfall in the trial site is mostly comprised of convective rain, but there is also a weak influence of monsoonal rain (Wong, 1989). Water balance and water discharge dynamics from the catchments 1985 - 1990 has been reported by Malmer (1992). Mean areal rainfall was 3215 mm and 3490 mm for W3 and W6 respectively during the five year period. For the same period mean catchment derived evapotranspiration was 1253 mm (W3) and 1540 mm (W6). The most humid year was 1988/89, the first year after treatments, with 4057 mm and 4340 mm areal rainfall for W3 and W6 respectively.

The research area includes two main types of vegetation. Three catchments (W1 - W3) comprised of forest selectively logged in 1981 and later struck by forest fire in 1982/83 (Woods, 1989). The remaining catchments (W4 - W6) comprised of dipterocarp forest lightly selectively logged in 1981.

Succession of vegetation

Biomass dry weight estimations based on destructive sampling before and after treatments were reported by Sim and Nykvist (1991). The hydrochemical monitoring started in November 1985, 2.5 years after the forest fire. At this time the above ground biomass at W1+2 was only 4.7 t ha^{-1} dw, mostly comprised of ferns, grasses and herbs. In late 1987, just before treatment, the biomass had increased to 26.7 t ha^{-1} dw of which 20.2 t ha^{-1} was accumulated in small pioneer trees. Dead trees are not included in these figures. No estimation of biomass in W3 and W6 was done. However, pre fire vegetation in W3 was not as damaged by the forest fire as W1+2. The areal extent of the forest fire was for W1+2 100 % and in W3 80 %. In W3, in a gradient from the water divide between W2 and W3 and to the fire boundary, there were more surviving trees from the pre fire period and less pioneer vegetation. Above ground biomass in W4 and W5 before treatment was 276 and 245 t ha^{-1} dw respectively at the time for clear-felling.

The accumulation of above ground biomass in tress and ground vegetation in the plantations after 1.5 years were in W1+2: 2.3, 6.2; W4: 10.5, 1.8 and W5: 5.4, 3.3 t ha^{-1} respectively (Sim and Nykvist, 1991). These differences were maintained and even more pronounced after 3.5 years in 1991 (Nykvist, pers. comm.).

MATERIALS AND METHODS

Forestry treatments

The treatments of the watersheds were as follows:

W1 Non-mechanized clearing of remaining trees and secondary vegetation, no wood extraction, burning of all biomass and planting.

W2 The same treatment as W1.

W3 Control catchment for forest fire area. No treatments.

W4 Manual felling, manual wood extraction, clearing of planting rows in the slash and planting in these rows, without burning.

W5 Manual felling, wood extraction using crawler tractors, burning of the remaining biomass and planting (normal practice).

W6 Control catchment for selectively logged area. No treatments.

Catchment W1 and W2 were treated as one because of uncertainty in identifying the phreatic water divide between the two streams. Malmer and Grip (1990) described the wood extraction methods used in this study in detail and resulting effects on soil physical properties.

Malmer and Grip (1990) reported the areal disturbance of top soil to 24% and 4% for tractor extraction and manual extraction respectively. The normal practice of burning remaining biomass before planting was used in W1+2 and W5. About one month after burning, grasses and weeds started to colonize the bare soil, and after three months most soil except for tractor tracks was covered with vegetation. At W4 burning was avoided and all remaining biomass was cut into smaller constituents and put into rows, leaving strips of bare ground accessible for planting. The planted tree species was *Acacia mangium* for all treated catchments.

Felling and wood extraction was carried out November 1987 to January 1988. Burning and rowing was done in February to April, and planting was conducted at W1+2 and W4 in March and W5 in May 1988.

Field sampling, laboratory handling and chemical analysis

A total of 2365 streamwater samples (including samples from 66 storm events) and 21 rainwater samples were analyzed for the time period covered by this paper. Details on sampling, laboratory handling and chemical analysis of streamwater and rainwater samples was given by Grip et al. (*in press*).

Methods for measurements of waterstage in the catchment streams and the computation of streamflow has been reported by Malmer (1992).

Data analysis

Regression analysis between concentrations and streamflow at sampling was made. These regressions were seldom significant and if so had low strength, except for some elements during and after treatments. Therefore regression analysis with dummy technique (Malmer, 1992, after Gujarati, 1978) could not be used to detect differences in concentrations between streams. Instead differences between mean concentrations between different streams during 5 periods were tested using Tukey-Kramer multiple test (Kramer, 1956) following tests of homogeneity in variance. The testing included differences between means of baseflow and stormflow in every stream and every period. The 5 periods used for the evaluation of the data here are the ones used by Malmer (1990 and 1992) with slight modifications, namely:

1/ reference period December 1985 - November 1987
2/ clear-felling and extraction period November 1987 - February 1988
3/ burning and plantation period March 1988 - July 1988
4/ first hydrologic year after treatments August 1988 - July 1989
5/ second hydrologic year and further August 1989 - December 1990

For period 5 the data on discharge and net loss of elements only cover August 1989 - July 1990 due to availability of streamflow data at the time of calculation.

To get a measure of association between the different elements, a correlation analysis (Wilkinson, 1990) was performed for the first year after treatment for treated streams (W1, W2, W4 and W5) and control streams (W3 and W6). Only sampling dates with complete chemical analysis were used. This gave sample sizes of about 80 for the whole dataset and about 25 for the sub-periods. These datasets only contained baseflow samples.

Methods for computation of the discharge of dissolved elements in streamwater and chemical input by bulk precipitation was given by Grip et al. (*in press*). The risk for underestimation of nutrient losses by unrecorded deep leakage from small headwater catchments has been discussed by Bruijnzeel (1990). Malmer (1992) discussed the possibility of this problem appearing in this study, and based on evapotranspiration, shallow soils and low soil permeability Grip et al. (*in press*) concluded it not to be a problem for the computation of element discharge.

Effects of element discharge due to treatments was calculated by regression analysis. Regressions between monthly dissolved discharge of elements were established between W3 to W1+2 and between W6 to W4 and W5 during the reference period. For these regressions n were about 20 and R^2 generally between 0.9 - 0.6. These regression equations were used to model what the discharge of elements would have been from the treated catchments if no treatments would have been applied. An approximation of monthly effects was calculated as the difference between modelled and measured monthly discharge of dissolved elements. Assuming that there were no changes in input trough wet deposition or weathering, these calculated effects were the sole net effect of the different treatments. Data given here does not include gaseous input or output of elements like nitrogen or particulate chemical input and losses in streams.

RESULTS

Streamwater element concentrations during reference period

Baseflow mean concentrations for different streams and different periods are tabulated by Grip et al. (Appendix 1, *in press*). Levels of element concentrations were low for all streams during the reference period. In spite of the low concentration levels there were differences in baseflow mean concentrations between streams during the reference period. The largest differences were found between the control streams W3 and W6. The other streams always had intermediate levels of concentrations, pH and electrical conductivity. Catchments W3 and W6 both have 100 % areal difference in soil type as well as different vegetation, while the others have an areal mix of the two soil types. These soil differences and local differences in bedrock in W3 were concluded to be the main reason for higher input from weathering to W3 and the resulting differences in streamwater concentrations (Grip et al., *in press*).

Streamwater element concentrations during and after treatments

In the control streams and treated streams before treatments, concentration levels of most elements were consistent troughout the 5 years of measurement, while the effect of treatments was often very clear on element concentrations in the treated streams. Monthly baseflow mean electrical conductivity clearly shows the response to clear-felling and burning (Figure 1). Some examples of

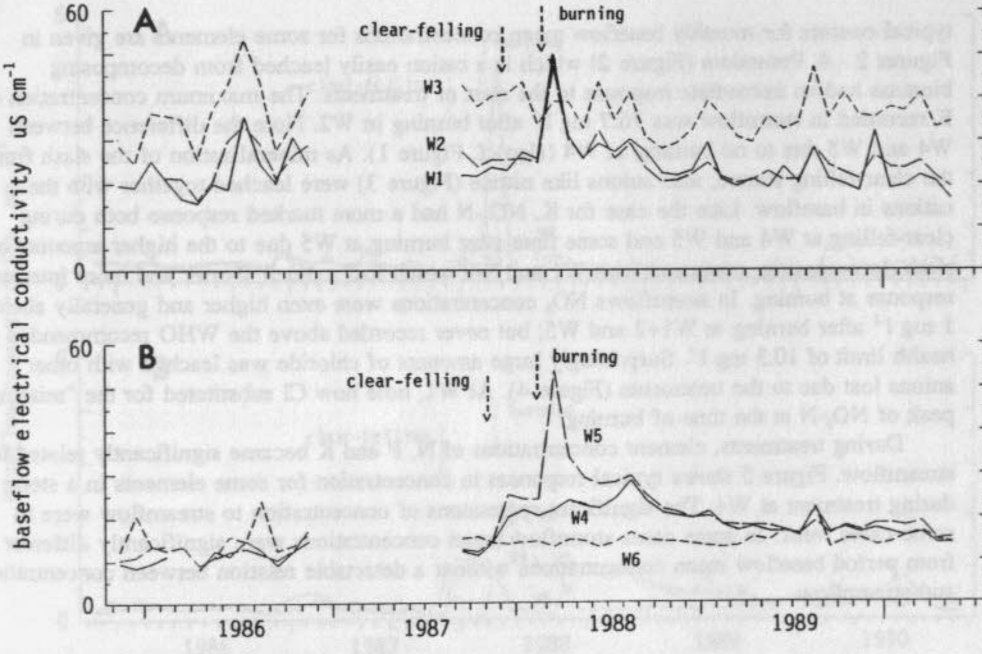

Figure 1. Monthly baseflow mean electrical conductivity (uS cm^{-1}) during November 1985 to May 1990 for catchments W1, W2 and W3 (A) and W4, W5 and W6 (B). See text for treatments.

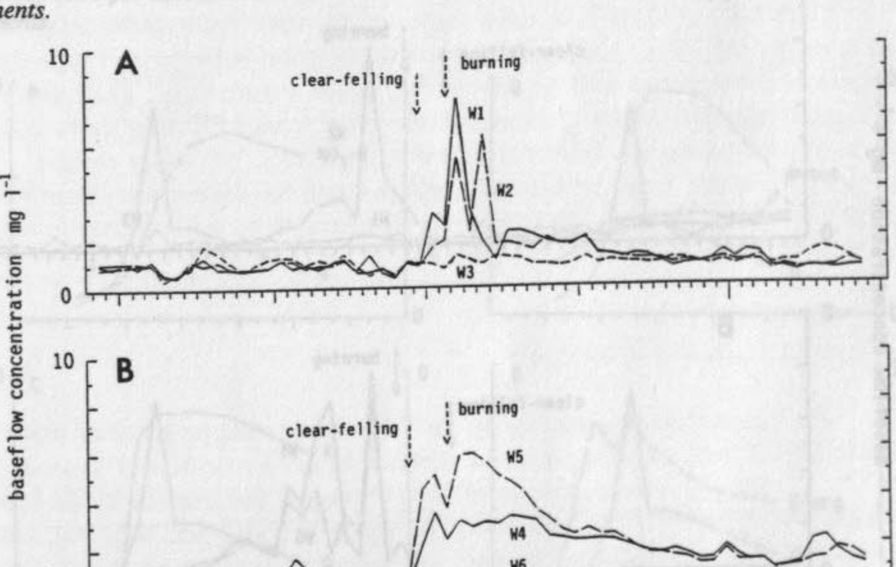

Figure 2. Monthly baseflow mean potassium concentration (mg l^{-1}) during November 1985 to November 1990 for catchments W1, W2 and W3 (A) and W4, W5 and W6 (B). See text for treatments.

typical courses for monthly baseflow mean concentrations for some elements are given in Figures 2 - 4. Potassium (Figure 2) which is a cation easily leached from decomposing biomass had an immediate response to the start of treatments. The maximum concentration of K recorded in stormflow was 76.7 mg l^{-1} after burning in W2. Note the difference between W4 and W5 due to no burning in W4 (also cf. Figure 1). As mineralization of the slash from the clear-felling started, also anions like nitrate (Figure 3) were leached together with the cations in baseflow. Like the case for K, NO_3-N had a more marked response both during clear-felling at W4 and W5 and some time after burning at W5 due to the higher amounts of slash decomposing, compared with W1 and W2 which had a much shorter and more intense response at burning. In stormflows NO_3 concentrations were even higher and generally above 1 mg l^{-1} after burning in W1+2 and W5, but never recorded above the WHO recommended health limit of 10.3 mg l^{-1}. Surprisingly large amounts of chloride was leached with other anions lost due to the treatments (Figure 4). At W1, note how Cl substituted for the "missing" peak of NO_3-N at the time of burning.

During treatments, element concentrations of N, P and K became significantly related to streamflow. Figure 5 shows typical responses in concentration for some elements in a storm during treatment at W4. The significant regressions of concentration to streamflow were in some cases weak. In some cases stormflow mean concentrations were significantly different from period baseflow mean concentrations without a detectable relation between concentration and streamflow.

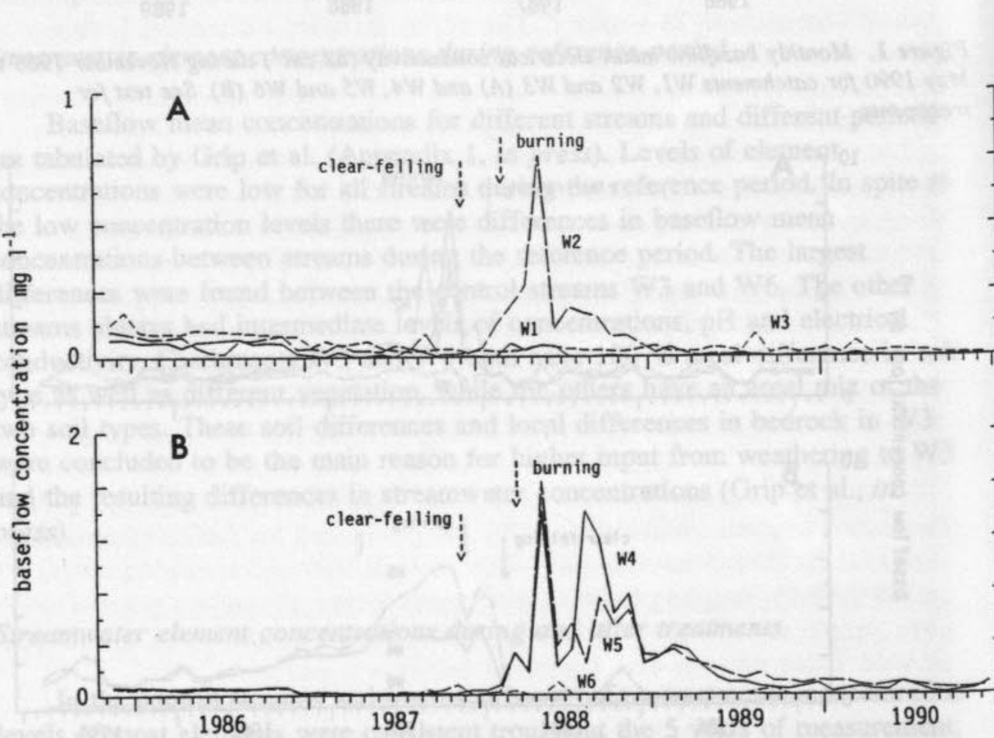

***Figure 3.** Monthly baseflow mean nitrate-nitrogen concentration (mg l^{-1}) during November 1985 to December 1990 for catchments W1, W2 and W3 (A) and W4, W5 and W6 (B). See text for treatments.*

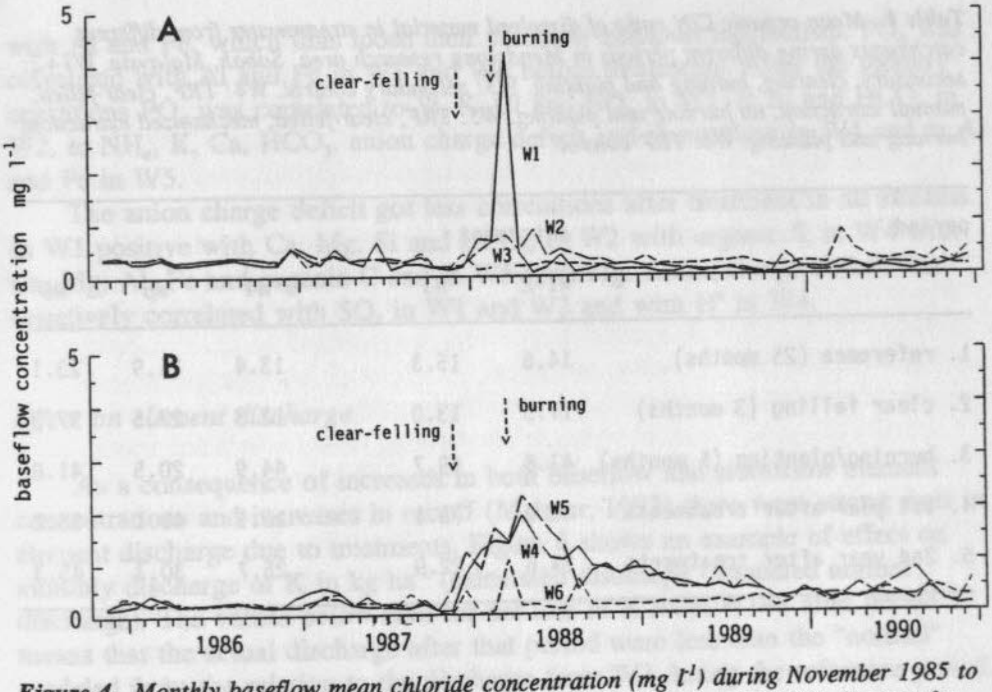

***Figure 4.** Monthly baseflow mean chloride concentration (mg l⁻¹) during November 1985 to December 1990 for catchments W1, W2 and W3 (A) and W4, W5 and W6 (B). See text for treatments.*

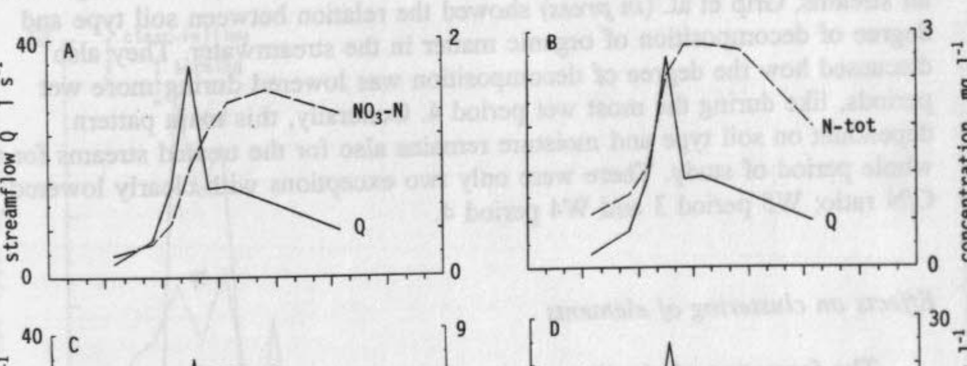

***Figure 5.** Response of dissolved nitrate-nitrogen (A), total nitrogen (B), potassium (C) and organic carbon (D) concentrations during a storm hydrograph in W4 after clear-felling at time for planting (3 March 1988).*

Table I. *Mean organic C/N ratio of dissolved material in streamwater from different catchments during different periods in Mendolong research area, Sabah, Malaysia. W1+2: secondary, clearing, burning and planting. W3: secondary control. W4: TRF, clear-felled, manual extraction, no burning and planting. W5: TRF, clear-felled, mechanized extraction, burning and planting. W6: TRF control.*

period	catchment				
	W1+2	W3	W4	W5	W6
1. reference (25 months)	14.6	15.3	13.4	21.9	23.1
2. clear felling (3 months)	17.5	13.0	12.8	27.5	37.3
3. burning/planting (5 months)	47.8	59.7	44.9	20.5	41.8
4. 1st year after treatments	45.0	79.1	15.1	40.1	55.2
5. 2nd year after treatments	24.6	22.9	25.7	30.3	45.3

Table I shows baseflow mean organic C/N ratio during different periods for all streams. Grip et al. (*in press*) showed the relation between soil type and degree of decomposition of organic matter in the streamwater. They also discussed how the degree of decomposition was lowered during more wet periods, like during the most wet period 4. Generally, this main pattern dependant on soil type and moisture remains also for the treated streams for the whole period of study. There were only two exceptions with clearly lowered C/N ratio; W5 period 3 and W4 period 4.

Effects on clustering of elements

The formation of significant ($p <= 0.01$) clusters increased after treatment, especially in W4, where streamflow entered the main cluster and was correlated to Ca, K, NH_4, PO_4, anion charge deficit, HCO_3, Cl and organic N. Si lost all correlations in W4 but was correlated to Mg in W1, W2 and W5. Mg, that was a central ion in W4 before treatment was only correlated to organic S after. In W2 and W5 the correlation between Mg and organic S was strengthened.

The Fe-Al-correlation remained strong in W4 and W5, but was broken in W1 and W2. The reason may be the relatively high concentration of basic cations after burning in W1 and W2 compared with W4 and W5 and the tendency of especially Ca and Mg to form complexes with organic anions in competitic with Al and Fe, which thus loose their common transport mechanism. PO_4 was correlated with Al and Fe in W1 and W4 before treatments. After the forestry operations PO_4 was correlated to Si, K, Cl and NO_3 in W1, to Si and HCO_3 in W2, to NH_4, K, Ca, HCO_3, anion charge deficit and streamflow in W4 and to Al and Fe in W5.

The anion charge deficit got less correlations after treatment in all streams. In W1 positive with Ca, Mg, Si and HCO_3, in W2 with organic S, in W4 with Ca, Mg, Al, Fe and organic C and in W5 with Mg. Anion charge deficit was negatively correlated with SO_4 in W1 and W2 and with H^+ in W4.

Effect on element discharge

As a consequence of increases in both baseflow and stormflow element concentrations and increases in runoff (Malmer, 1992) there were strong rises in element discharge due to treatments. Figure 6 shows an example of effect on monthly discharge of K in kg ha^{-1} (calculated discharge - modeled normal discharge). The values below zero for net loss of K from W1+2 after period 3 means that the actual discharge after that period were less than the "normal" modeled from the relation to the discharge from W3 during the reference period. That effect could therefore be called a "reduced loss".

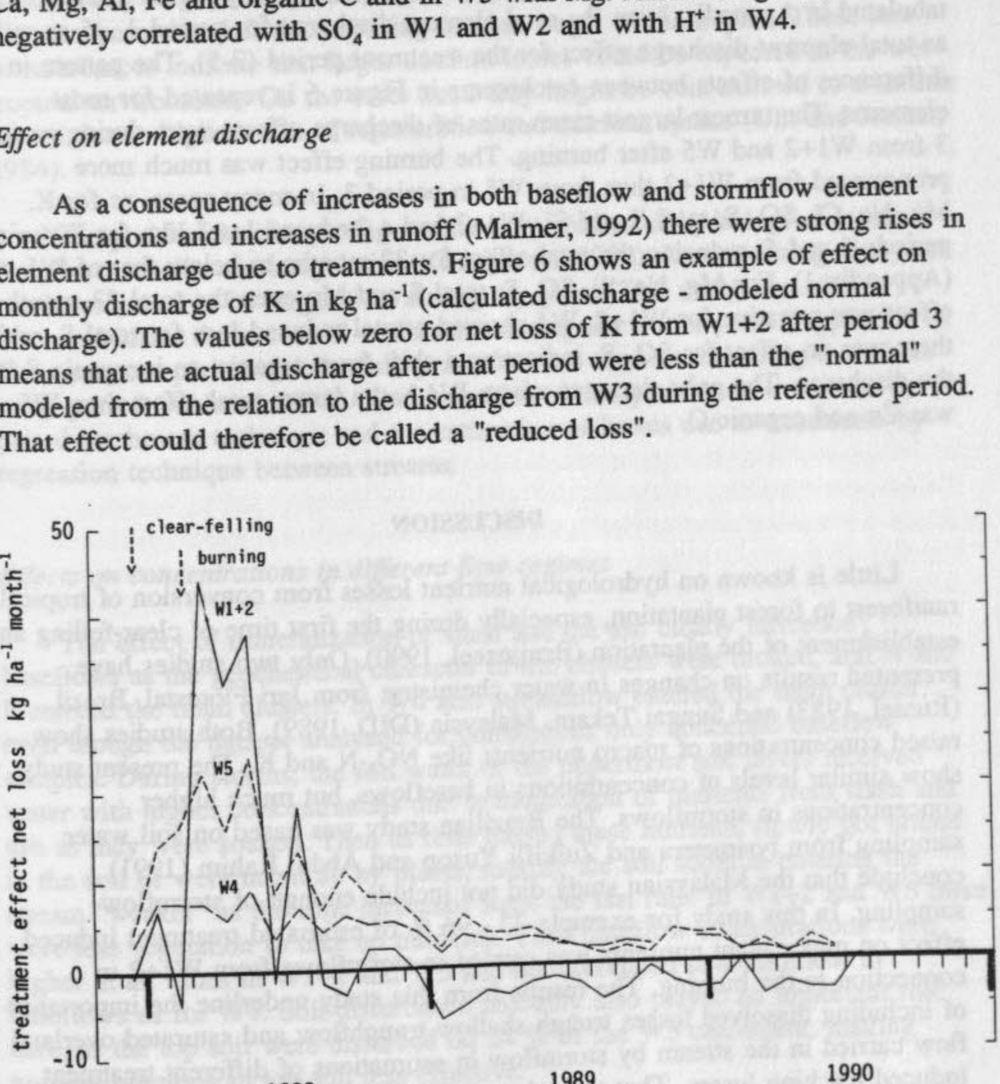

Figure 6. *Effect of treatments on losses of dissolved potassium (kg ha^{-1} mon^{-1}) for catchments W1+2, W4 and W5. Se text for treatments.*

The greatest effect was recorded from W1+2 at and a few months after burning. W5 had a higher effect during clear-felling and extraction than W4 and a marked maximum during and after burning. W4 had a lower effect during clear-felling and extraction than W5 and no extra effect due to burning, resulting in a much lower effect during period 2 and 3. The effect from W4 and W5 was still recognizable in mid 1990, 33 months after start of treatments.

The mean rate of dissolved element discharge (kg ha^{-1} $month^{-1}$) during the different periods for the five catchments are tabulated in Appendix 1. Also tabulated in Appendix 1 are the total element discharge for period 1 - 5 as well as total element discharge effect for the treatment period (2-5). The pattern in differences of effects between catchments in Figure 6 is repeated for most elements. The utmost largest mean rates of discharge effects were during period 3 from W1+2 and W5 after burning. The burning effect was much more pronounced from W1+2 than from W5 in period 3. In many cases, as for K, Mg, Na, Cl, SO_4-S, total S and Si, W1+2 had a "reduced loss" like for K during periods 4 and 5, reducing the total effect for 33 months to below that of W5 (Appendix 1). For Mg, Na, Si, SO_4-S, total S and Mn even the total 33 month effect was negative for W1+2. W4 showed a total reduced loss for total S, while there was an effect for SO_4-S, indicating a shift from organic- to inorganic S in the discharge. The only elements where W4 had a larger total effect than W5 was Zn and organic C.

DISCUSSION

Little is known on hydrological nutrient losses from conversion of tropical raniforest to forest plantation, especially during the first time of clear-felling and establishment of the plantation (Bruijnzeel, 1990). Only two studies have presented results on changes in water chemistry; from Jari Florestal, Brazil (Russel, 1983) and Sungai Tekam, Malaysia (DID, 1989). Both studies show raised concentrations of macro nutrients like NO_3-N and K. The present study show similar levels of concentrations in baseflows, but much higher concentrations in stormflows. The Brazilian study was based on soil water sampling from lysimeters and Zulkifli Yusop and Abdul Rahim (1991) conclude that the Malaysian study did not include enough of stormflow sampling. In this study for example 73 - 96 % of estimated treatment induced effect on major plant nutrients was carried in stormflows from W1+2 in connection to the burning. The results from this study underline the importance of including dissolved losses trough shallow troughflow and saturated overland flow carried in the stream by stormflow in estimations of different treatment induced leaching losses. That the sole use of lysimetry or baseflow sampling may lead to substantial underestimations has also been showed in many studies from other climates (Reynolds et al., 1990).

In temperate climate, exports of nutrients following clear-felling and plantation have been shown to be moderate (e.g. Bormann and Likens, 1967; Grip, 1982; Rosén, 1982) as for warm-temperate climates (e.g. Swank and Douglass, 1975; Hewlett et al., 1984; Hopmans et al., 1987). In many cases these increased nutrient exports have been mainly attributed to increased runoff (e.g. Hewlett et al., 1984; Swank, 1988). In the wet tropics rainfall surplus over evapotranspiration is larger and runoff increases due to clear-felling can be considerably larger (Bruijnzeel, 1990). Also, like in this study, the amount of biomass burned or left decomposing is often larger in the tropics. These two circumstances indicate that larger nutrient losses could be expected in the wet tropical environment. On the other hand they might be concentrated to a shorter period due to higher rates of regeneration and nutrient uptake (Uhl and Jordan, 1984).

One problem in interpreting results from studies of hydrological losses is the effects of climatic irregularities (Bruijnzeel, 1990). Losses during wet years can be assumed to be larger than during dry years. This study also shows increased losses from control catchments during the wetter years (Appendix 1). The treatment period and the first year after was the wettest during the whole study period (Malmer, 1992). The main bias of this problem is overcome by the paired catchment technique and the estimation of losses due to treatments by regression technique between streams.

Effects on concentrations in different flow regimes

The effect of mineralization of slash and the ash clearly influence baseflows as the geochemical character of the clusters were broken, and N and P entered the main clusters. In W4 also streamflow entered the main cluster, even though the dataset analyzed for correlations only contained baseflow samples. During storms, the soil water of the uppermost soil layers received water with higher concentrations due to dissolution of nutrients from slash and ash as they were soaked. Then as time passed these nutrients slowly got bound in the soil or were taken up by plants, making the soil solution entering the stream "weaker" as the time increased since the last rain. In W1+2 and W5 there were less vegetation to take up nutrients. Thus baseflow concentrations were higher at all times in W1+2 and W5 and not correlated to streamflow in baseflows as for W4. Soil disturbance probably also played an important role here, as the top soil were disturbed on 24 % of the W5 catchment, making nutrient retention in top soil less effective.

The enrichment of elements in stormflow compared to baseflow was almost only apparent during the treatment periods 2 and 3. This marks the time with the highest mineralization of nutrient rich green parts of the slash, and high amounts of available nutrients in the ash. At this period dissolved elements were easily carried away by overland flow and saturated overland flow to the stream during storms.

A reason for differences between streams in increased concentrations due to treatments, apart from amount of biomass mineralized and burned, might be the differences in effect of treatments on streamwater discharge dynamics. Malmer (1992) reported significantly reduced streamwater baseflow and increased stormflows for W1+2. W5 had the opposite change in discharge dynamics compared to W1+2, while W4 had no significant changes in discharge dynamics during treatments compared to control. This coincided well with W1+2 having the most extreme maximum concentrations during stormflows, while W5 had the clearest treatment induced concentration increases in baseflow (cf. Figures 1 and 2).

The rather short lived effect of raised concentrations for most elements coincides well with earlier findings (Russel, 1983; Zulkifli Yusop, 1989). Within less than two years NO_3-N for example was back to pretreatment levels for all studies. Some elements made exceptions, like K with still elevated concentrations after 5 years (Zulkifli Yusop, 1991) and 3 years (this study). K is one of the most easily leached ions, and as Zulkifli Yusop (1991) pointed out, weathering might be enhanced by increased soil temperature and moisture after treatment. Also it will take a longer time for the growing plantation to develop deeper root systems to take up elements from weathering in deeper soil layers. Consequently the faster decrease in nitrate leaching indicate the high rate of biomass recovery taking up surficially mineralized nutrients mainly emanating from pretreatment biomass, and the prolonged higher levels of potassium indicate losses of nutrients derived from weathering (c.f. Figures 2 and 3). Another reason for prolonged high concentrations of K could be that some water enriched by leaching from biomass did have a deeper flow pathway and thus a longer transit time. However, the fact that NO_3 and NH_4, only apparent in biomass and not in minerals, not have the same pattern of prolonged raised concentrations as the basic cations, supports the hypothesis that the short lived peaks of different nutrients leached mainly indicate loss from biomass and the more prolonged higher levels of concentrations mainly emanate from weathering.

Effects on element discharge

The nutrients carried as effect of treatments in streamwater has two major sources; either mineralization of the slash trough decomposition and soluble nutrients by burning or weathering of soil and bedrock. Element retention can be expected to be dependant on amount of living biomass left and the degree of soil disturbance. The more vegetation killed, the less nutrients will be retained by uptake in living biomass in the earliest stage. Soil disturbance and especially removal of top soil reduces capacity for retention in the soil by reducing the biomass of the soil fauna (Jordan, 1991) and by loss of exchange capacity. Also, both less vegetation left and soil disturbance increases runoff and surficial drainage (Malmer, 1992). Consequently, as shown in this study, losses were greatest from the combination of both burning and soil disturbance. On the other hand it may be assumed that the total loss from repeated burning, like W1+2, might have been even larger. Silvicultural methods must be adapted to reduce soil disturbance and leave living biomass. Especially leaving trees would be effective as they by deeper root systems also might reduce losses from weathering. Hopmans (1987) ascribed lack of leaching losses after clear-felling and burning in south-eastern Australia to a 30 m wide buffer strip along the stream.

Some parts of the elements released by weathering may not be active in the biological part of the nutrient cycling if soils are deep and strongly weathered (Baille, 1989; Bruijnzeel, 1989). As soils in this study are comparatively shallow and losses from control catchments comparatively low, the biologically inactive contribution of weathering is probably rather small in the catchments as long as they are under forest cover. After clear-felling this contribution to the element discharge might rise as very few deep rooted plants remain to utilize weathered minerals on deeper soil levels. It is not possible to make anything else than hypothetical assumptions on differences between contribution from weathering or mineralization of biomass in the discharge at this stage (c.f. concentrations above). Anyway the discharge effect induced by treatment, as calculated here, was mainly consisting of elements that could have been in reach of the biologically active part of the ecosystem and which represent a lost capital of nutrients for future biomass production.

As a comparison the discharge effect of elements from W5 and the contents in easily mineralized parts of the biomass (leaves, twigs, fine roots, and ground vegetation, Sim and Nykvist, 1991) are tabulated in Table II. The percentage for most elements are around 10 % discharge effect compared to the contents of "easily mineralized" biomass. The exception is K and total S which are lost to around 80 % of the content in the compared biomass. For these two elements the losses as discharge effect was larger than the contents in stems with bark in

Table II. Content of some elements (kg ha^{-1}) in the most easily decomposed parts of the biomass (after Sim and Nykvist, 1991) compared to the amount of the same elements discharged in streamwater as an effect of clear-felling, mechanized extraction and burning during 9 months of treatments (period 2 - 3) and for the total period of 33 months (period 2-5) during and after treatments. Catchment W5 in Mendolong research area, Sabah, Malaysia.

element	tree leaves	branches < 2 cm	other[1] veg.	roots < 2 cm	total	discharge per 2-3	effect per 2-5
N-tot	89	48	29	204	370	13	40
P-tot	4.1	1.5	1.3	7.0	13.9	0.9	1.3
K	61	41	21	88	211	94	181
Ca	57	65	5	73	200	12	27
Mg	20	11	4	32	67	8	15
S-tot	9	6	3	20	38	20	31
Mn	3.2	1.4	0.8	2.7	8.1	0.4	0.6

1. All other above ground biomass excluding trees with diameter > 20 cm.

the clear-felled rainfores (Sim and Nykvist, 1991). These figures are remarkably high. By also adapting the hypothesis above, on the losses from weathering on deeper soil layers becoming apparent in the prolonged loss of some elements like K, it could be concluded from Figure 6 that the bulk of the losses were during the first year and emanate from ash and mineralized slash.

Effects related to burning

Different kinds of burning are used in the tropics for a wide range of site preparations and forest plantation management. Many types of prescribed burning in forest plantations fulfill goals of increasing soil nutrient availability and suppressing weeds (de Ronde et al., 1990). These kinds of fires are mostly sweeping fires only burning off ground vegetation, leaving trees and soil organic matter intact and keeping soil temperature down during burning. On the other hand some results from burning logging residues in Australia indicate the risk for long term depletive effects on soil nutrients and plantation productivity (eg. Flinn, 1981; Turvey and Cameron, 1986). Peters and Neuenschwander (1988) show the example of how many traditional shifting cultivators avoid areas with harder burning. The harder and hotter burn results in whiter ash for example where stumps or biomass piles burn for a longer time. Less black ash is indicative of lower carbon content and higher concentrations of nutrients, and the loss of nutrients in smoke is proportional to the amount of weight loss of fuels (Raison et al., 1985). In the case of this study the normal practice for the area of burning the slash after clear-felling was used. In this region the amount of fuel in remaining biomass after clear-felling is very high, and one of the main reasons for burning is to get the soil accessible for planting. Because of this, harder burning is preferred, where not only leaves, twigs and ground vegetation, but also wood in branches, rejected logs and stumps might burn for a long time. This practice often leave a very mixed pattern of areal distribution of patches with differently burned slash and soil. The often irregular burning pattern increases an already high variability of nutrient availability of the soil (Lal et al., 1975).

As discussed above the effect of burning was also very clear for both baseflow and stormflow concentrations in period 3. This effect on concentrations induced by burning was short lived, between 1 to 6 months. It also produced the most extreme concentrations recorded, like regular concentrations of above 1 mg l^{-1} of NO_3-N in stormflow samples, especially in W1+2. At W5 this effect in stormflow during period 3 was weaker for total N, Ca, Cl and electrical conductivity, probably due to the more marked increase i baseflow discussed above as well as a less hard burn in W5.

The "reduced leaching" for K (Figure 6) and some other elements in W1+2 could have had two main reasons. One reason might be that W1+2 in 1985 - 1987 still was in a higher recession in losses than W3 due to the forest fire in 1982/83. There is no sign of that as for concentrations (eg. Figure 2). Rather W3 had higher concentrations and losses for most elements during the reference period. On the other hand W1+2 had a stronger recession of runoff during the reference period (Malmer, 1992). The other possible reason for the negative effect might be exhaustion of leachable elements in W1+2 after burning. W1+2 had probably a substantial loss of nutrients already after the forest fire, as the fire killed all larger trees and most vegetation (cf. above). Again the second burning in 1988 resulted in further substantial losses. This second burning was harder and "more successful" than the one in W5 as to converting biomass material to ash. Consequently the loss in connection to burning is double the loss from W5 (Figure 6), even though the total amount of biomass fuel was much larger in W5. After this second burning there were probably much less easily leached elements in ash and soil in W1+2 than in W5 making up the "exhaustion effect" compared with the leaching from the secondary vegetation during the reference period. The much slower growth of the planted trees in

W1+2 (cf. above) also indicate that a deficiency of nutrients might be at hand, reducing leaching losses to below pretreatment levels as trees and weeds start to grow in period 4.

The fact that PO_4-P became correlated to Si in W1 and W2 may be connected to that the burning of W1+2 was harder than in W5, making the link between the two more apparent in the ash. That also indicates that particulate losses of P in ash carried as suspended load might be more apparent from W1+2 than W5.

CONCLUSIONS

Effects on streamwater concentrations due to leaching during treatments were the lowest with manual extraction and no burning before planting due to; 1) more living plants left to take up nutrients and use water, 2) low disturbance to soils and streamflow discharge dynamics and 3) no burning making release of mineralized elements slower.

For sustainable forest production in natural rainforests, as well as in plantation forestry, it is of great importance to minimize nutrient leaching in connection with forestry operations. After clear-felling, streamflow increases along with the concentration of essential nutrients. The normal practice of burning add to the loss due to clear-felling, and should be avoided. Extensive soil disturbance also add to dissolved losses and have to be kept to a minimum. Leaving some vegetation like ground vegetation and riparian reserves makes possibilities to contain nutrients from leaching larger.

With the high losses of nutrient described here, it becomes very important to quantify the rate of weathering to access possibilities for long term productivity trough several generations of forest plantations. Very little is known on weathering rates under forests on different soil types and bedrock in the tropics. The balance of weathering and mineralization to the uptake of trees and other vegetation in different stages of silviculture is still to a high degree a black box.

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Appendix 1.

Mean rate of dissolved element discharge in streamwater (kg ha^{-1} mon^{-1}) for reference period (1) and periods during (2,3) and after (4,5) treatments. The total dissolved discharge for period 1 - 5 (58 months) is also tabulated together with the total effect (total discharge - modelled control discharge) for period 2-5 (33 months) (kg ha^{-1}).

element/ /period	catchment stream W1+2	W3	W4	W6	W5
NO_3-N					
1	0.06	0.06	0.05	0.05	0.06
2	0.15	0.09	0.30	0.02	0.32
3	0.60	0.05	0.57	0.01	0.60
4	0.06	0.06	0.24	0.02	0.57
5	0.01	0.04	0.11	0.01	0.26
tot discharge	4.4	1.8	7.9	0.5	13.9
tot effect	3.0		7.0		12.7
NH_4-N					
1	0.01	0.01	0.01	0.01	0.02
2	0.04	0.05	0.26	0.03	0.47
3	0.48	0.02	0.24	0.02	0.26
4	0.06	0.04	0.07	0.04	0.07
5	0.06	0.05	0.10	0.03	0.05
tot discharge	3.9	1.4	4.0	1.0	4.1
tot effect	2.1		2.9		3.3
N-tot					
1	0.39	0.33	0.47	0.49	0.52
2	0.32	0.32	0.82	0.57	1.29
3	3.12	0.38	1.89	0.87	2.76
4	0.81	0.35	1.84	0.75	2.04
5	0.48	0.42	0.78	0.49	1.27
tot discharge	32.8	12.1	43.4	20.9	57.5
tot effect	17.2		27.0		39.9
PO_4-P					
1	0.01	0.02	0.01	<0.005	0.01
2	0.01	0.01	0.02	0.01	0.07
3	0.21	0.01	0.04	0.01	0.10
4	0.02	0.04	0.02	0.01	0.02
5	0.03	0.03	0.02	0.01	0.03
tot discharge	1.7	0.9	0.7	0.3	1.2
tot effect	1.0		0.2		0.6
P-tot					
1	0.02	0.02	0.02	0.01	0.01
2	0.03	0.02	0.03	0.01	0.10
3	0.23	0.02	0.05	0.01	0.13
4	0.06	0.06	0.04	0.02	0.05
5	0.04	0.04	0.04	0.02	0.04
tot discharge	2.4	1.4	1.2	0.7	2.0
tot effect	1.8		0.8		1.3

element/ /period	catchment stream W1+2	W3	W4	W6	W5
K					
1	1.96	1.24	0.76	0.57	0.20
2	5.49	1.84	6.95	0.44	12.27
3	32.21	1.73	7.55	0.55	17.49
4	4.22	2.14	4.68	0.56	6.27
5	1.64	1.50	2.28	0.36	2.55
tot discharge	247.8	57.8	142.2	15.1	230.2
tot effect	83.9		105.8		189.4
Ca					
1	1.96	2.64	0.43	0.24	0.51
2	2.04	3.39	0.90	0.29	1.97
3	6.07	2.53	1.88	0.43	2.11
4	3.57	3.52	1.07	0.76	1.68
5	1.83	2.22	0.61	0.21	1.01
tot discharge	101.2	91.7	32.3	14.6	48.7
tot effect	28.2		25.0		26.7
Mg					
1	1.46	2.13	0.52	0.20	0.44
2	1.45	3.13	1.12	0.34	1.75
3	1.89	2.32	1.55	0.70	2.24
4	2.06	2.84	1.09	0.71	1.32
5	1.20	2.14	0.64	0.17	0.87
tot discharge	52.9	80.8	32.0	15.0	42.7
tot effect	-5.1		7.7		15.5
Na					
1	1.98	3.88	1.62	0.88	1.76
2	2.20	5.36	2.35	1.22	2.26
3	2.04	3.42	1.58	0.76	2.24
4	1.90	4.36	1.52	0.88	1.73
5	2.50	4.83	1.89	1.26	2.77
tot discharge	69.7	143.3	55.9	33.1	72.0
tot effect	-9.4		1.2		9.9
Si					
1	2.24	4.18	0.74	0.45	0.76
2	3.28	6.63	2.88	2.26	3.19
3	3.75	6.09	2.79	1.00	4.13
4	3.37	8.76	1.01	1.03	1.35
5	2.64	5.94	0.89	0.39	1.40
tot discharge	100.7	226.9	45.4	28.8	63.4
tot effect	-17.5		8.6		16.8

element/ /period	catchment stream W1+2	W3	W4	W6	W5
Fe					
1	0.30	0.13	0.55	0.51	0.53
2	1.29	0.60	2.94	1.35	6.94
3	0.58	0.12	0.34	0.31	0.71
4	0.74	0.20	1.06	0.88	1.80
5	0.32	0.05	0.66	0.48	1.00
tot discharge	19.4	5.4	31.1	22.0	57.9
tot effect	10.0		22.0		38.0
Al					
1	0.08	0.04	0.11	0.36	0.22
2	0.33	0.22	0.88	0.77	4.77
3	0.28	0.05	0.21	0.46	0.64
4	0.22	0.06	0.24	0.54	0.56
5	0.08	0.03	0.14	0.31	0.24
tot discharge	5.9	1.9	8.2	14.8	27.1
tot effect	2.3		4.4		19.1
Cl					
1	0.42	0.41	0.30	0.31	0.32
2	1.57	0.41	2.74	0.25	3.06
3	2.43	0.31	2.80	0.58	3.93
4	0.48	0.76	1.56	0.46	1.74
5	0.20	0.81	0.55	0.41	0.81
tot discharge	25.0	21.6	47.5	14.1	59.5
tot effect	0.7		38.2		47.9
SO_4-S					
1	2.54	3.27	0.97	0.22	0.88
2	2.67	4.54	1.37	0.30	1.37
3	5.09	3.36	1.35	0.35	1.82
4	2.93	4.28	1.56	0.35	1.28
5	1.74	3.30	1.24	0.19	1.24
tot discharge	89.5	121.4	44.5	9.2	42.1
tot effect	-7.8		9.9		11.2
S-tot					
1	2.54	3.27	1.07	0.32	0.94
2	2.67	4.82	1.79	0.50	1.40
3	5.41	3.36	2.50	1.07	4.74
4	3.00	4.28	1.58	0.93	1.62
5	1.74	3.30	1.26	0.24	1.25
tot discharge	89.5	121.4	51.8	20.8	62.4
tot effect	-6.4		-17.2		30.6

element/ /period	catchment stream W1+2	W3	W4	W6	W5
Mn					
1	0.03	0.01	0.02	0.01	0.02
2	0.04	0.01	0.07	0.01	0.13
3	0.08	0.03	0.04	0.03	0.05
4	0.03	0.02	0.04	0.01	0.03
5	0.02	0.01	0.03	0.01	0.02
tot discharge	1.2	0.5	1.3	0.5	1.2
tot effect	-0.2		0.5		0.6
Zn					
1	0.01	0.01	0.01	0.01	0.02
2	0.10	0.02	0.02	0.02	0.04
3	0.04	0.03	0.04	0.06	0.09
4	0.03	0.01	0.11	0.03	0.04
5	0.06	0.03	0.04	0.04	0.06
tot discharge	1.6	0.8	2.1	1.2	1.8
tot effect	0.3		1.1		0.4
C-org					
1	2.69	1.08	2.80	5.49	4.59
2	15.62	4.92	7.92	20.61	22.64
3	21.64	13.10	26.62	32.31	39.08
4	36.95	21.54	25.18	36.84	35.67
5	10.78	5.16	14.30	19.91	23.80
tot discharge	727.9	400.8	629.9	904.4	976.4
tot effect	129.9		198.3		167.8

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OBSERVATIONS ON SLOPE HYDROLOGY AND EROSION IN TROPICAL RAINFOREST, AND AS RESPONSE TO CLEAR-FELLING, SOIL DISTURBANCE AND SUBSEQUENT BURNING IN SABAH, MALAYSIA.

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ABSTRACT

Plot studies of slope surface runoff and surface erosion, as well as measurements of field saturated hydraulic conductivity and observations of soil moisture were made during one year in control forest and in clear-felled experimental catchments during conversion to forest plantation. Gully erosion on tractor tracks were recorded during 3 years. In one catchment conversion soil disturbance at extraction and no burning of residues was practised before planting (W4). In the other catchment crawler tractors were used for extraction and residues were burned before planting (W5). The plots in the treated catchments were on soils undisturbed by wood extraction to separate effects of clear-felling with or without subsequent burning.

In the control forest transport of water down the slopes to the stream was concluded to occur mainly as throughflow in permeable top soils, with rainfall activating pre-event water by reactivation of highly conductive top soil pores. Only 2.9 % of yearly rainfall in a wet year occurred as slope surface runoff, and was attributed to infiltration excess overland flow during short periods of rains. Based on rapid decrease in hydraulic conductivity with depth, patterns of soil moisture and stream baseflow element concentrations, [illegible] was concluded to be supplied both by water from deeper soil horizons [illegible] throughflow. Saturation overland flow adjacent to the stream was contrib[illegible] stream stormflow.

Extensive slope overland flow did not occur in any investigated cl[illegible] soil disturbance. Earlier reported high dissolved element concentrations in stream stormflows after clear-felling was concluded to originate from soil water with high solute load leached from residues or ash, travelling as throughflow with short residence time in large top soil pores and from water in contact with residues or ash at surficial flows adjacent to the stream. Dissolved element leaching could be reduced by keeping living vegetation, minimizing soil disturbance and avoiding burning.

Recorded surface erosion on undisturbed soils after clear-felling on both W4 and W5 were in the range of undisturbed natural forests. However, despite small amounts, the quality of lost sediments is underlined, as the mass of applied [illegible] in the first 3 days of intensive rainfall after burning in W5 was double the yearly surface sediment loss in control forest.

Disturbed soil surfaces on tractor tracks, on the contrary, experienced extensive surface runoff as infiltration excess overland flow, causing severe gully erosion contributing to earlier reported stream siltation. Later, natural rehabilitation of the disturbed soils has shown to be very slow and contribute to prolonged increases in runoff and its stormflow component.

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In the control forest transport of water down the slopes to the stream was concluded to occur mainly as troughflow in permeable top soils, with rainfall activating pre-event water by resaturation of highly conductive top soil pores. Only 2.9 % of yearly rainfall in a wet year occurred as slope surface runoff, and was attributed to infiltration excess overland flow during short sub periods of rains. Based on rapid decreace in hydraulic conductivity with depth, patterns of soil moisture and stream baseflow element concentrations, even stream baseflow was concluded to be supplied both by water from deeper soil horizons and shallow top soil throughflow. Saturation overland flow adjacent to the stream was considered likely to supply stream stormflow.

Extensive slope overland flow did not occur in any investigated clear-felled area without soil disturbance. Earlier reported high dissolved element concentrations in stream stormflows after clear-felling was concluded to originate from soil water with high solute load leached from residues or ash, travelling as throughflow with short residence time in large top soil pores and from water in contact with residues or ash at surficial flows adjacent to the stream. Dissolved element leaching could be reduced by keeping living vegetation, minimizing soil disturbance and avoiding burning.

Recorded surface erosion on undisturbed soils after clear-felling on both W4 and W5 were in the range of undisturbed natural forests. However, despite small amounts, the quality of lost sediments is underlined, as the mass of eroded ashes in the first 3 days of moderate rainfall after burning in W5 was double the yearly surface sediment loss in control forest.

Disturbed soil surfaces on tractor tracks, on the contrary, experienced extensive surface runoff as infiltration excess overland flow, creating severe gully erosion contributing to earlier reported stream siltation. Later, natural rehabilitation of the disturbed soils has shown to be very slow and contribute to prolonged increases in runoff and its stormflow component.

INTRODUCTION

The resource of the managed natural forest and the use of extractive tree plantations are becoming increasingly important due to the depletion of the international forest resource in the tropics (Whitmore, 1990). Logging of tropical rainforest cause disturbance to soil and water. This disturbance will vary in amount and duration depending on intensity of the operation and the methods used (Hamilton and King, 1983). To secure sustainability of future forest production it will be increasingly important to optimize silvicultural systems, for selective logging and for land clearing for forest plantations, to minimize negative impacts on soil and water. Erosion and changes in hydrology have been studied for a long time on the plot scale as well as on the catchment scale. However, to understand the dynamics of several different combined disturbances (clear-felling, soil disturbance and burning) in different parts of a catchment and the effect on nutrient dynamics and site sustainable production, different approaches and scales have to be combined (Anderson and Spencer, 1991). Few such combined approach studies has been published from tropical forests (Bruijnzeel, 1990).

The flow path of water from its arrival at the soil surface to its delivery in the stream is decisive for the transport of eroded sediments (Douglas and Spencer, 1985) and the leaching, transport and losses of solutes from slopes (Pilgrim et al., 1979; Burt 1986). In the humid tropical environment, the importance of a bi-phasic flow regime of bulk slope throughflow with short transit time through large pores, in between aggregates, has been stressed as an important nutrient conserving mechanism (Northcliff and Thornes, 1978 and 1989; Sollins and Radulovich, 1988). At high rates of mineralization like after logging, clear-felling or burning the same mechanism may lead to extensive export of nutrients, as indicated by Malmer and Grip (*in press*). Chemical non equilibrium is also believed to be prevalent in mesopores during storm events (Luxmoore et al., 1990; Wilson et al., 1991). However, the complex interaction between solutes in different pore sizes and the organic and geochemical sources and stores of elements in surficial soil horizons is poorly known.

The objective of this paper is to summarize observations on soil physical properties, hydrological conditions and erosion to increase the understanding of the mechanisms behind catchment scale effects of some forestry treatments applied in this study. This paired catchment experiment includes comparisons of different wood extraction systems and preplanting practices when converting tropical rainforest to forest plantation. Reports on treatment effects have been presented elsewhere (Malmer and Grip, 1990 and *in press*; Malmer, 1990 and 1992; Sim and Nykvist, 1991; Grip and Malmer, *in press*). Observations on slope surface runoff, soil moisture and surface and gully erosion are presented in this paper and put together with data from earlier papers to discuss the water and soil physical interactions after the different forestry treatments.

Finally a short discussion on possible differences in effects between clear-felling and selective logging is made. Selective logging of tropical rainforests is today practiced over much larger areas than conversion to extractive tree crop plantations, but soil disturbance and the effects on hydrology and erosion are similar in many aspects to those from clear-felling and conversion to forest plantation.

RESEARCH AREA

The Mendolong research area including the six catchments used for this impact study is situated at 650 - 750 m.a.s.l. on the foothills of Mount Lumako in the Crocker Range 35 km south east of the coast at Sipitang (115.5° E,5.0° N) in Sabah, Malaysia. Location and layout of the catchments can be seen in Figure 1.

The natural vegetation of the research area is a lowland hill dipterocarp forest (Whitmore, 1984, after Symington, 1943), although the altitude is just below the transition to lower montane forest. The forest of the research area was lightly selectively logged in 1981 and a forest inventory and biomass investigation was published by Sim and Nykvist (1991). Three of the catchments were struck by the "Borneo forest fire" in 1982/83 (Woods, 1989) and comprised at the time of this study of secondary vegetation (Sim and Nykvist, 1991). The size of the catchments varies from 3.4 to 18.2 ha. Topography of the area is mostly moderately sloping, ranging up to 27 %, but steeper slopes of up to 57% occur.

The top soil in the research area was an areal mix of clay soil (40% clay, 40% sand) and loamy sand soil (10% clay, 80% sand) (Malmer and Grip, 1990). The areal distribution of the two soil types were reported by Grip et al. (*in press*). Dry bulk densities and steady state infiltrabilities of the two different top soils were presented by Malmer and Grip (1990). They also reported the change in these properties on disturbed soils after different wood extraction methods as well as the areal disturbance from these methods and some data on porosity of top soil.

Mean areal rainfall for the two control catchments W3 and W6 were 3215 and 3490 mm respectively for the first five year period of measurement 1985/86 - 1989/90. Mean yearly runoff for the same period and catchments were 1962 and 1950 mm respectively (Malmer, 1992). Yearly increases in runoff due to treatments were also reported by Malmer (1992). Soil loss through stream suspended sediments from control catchments were in the range of 0.2 to 0.4 t ha^{-1} y^{-1} (Malmer, 1990). Stream suspended load output from different catchments during different periods of treatments were also reported by Malmer (1990). Streamwater dissolved losses from the catchments during different periods were reported by Grip et al. (*in press*) and Malmer and Grip (*in press*). Streamwater dissolved loss from control catchments were in the order of 0.2 t ha^{-1} y^{-1}.

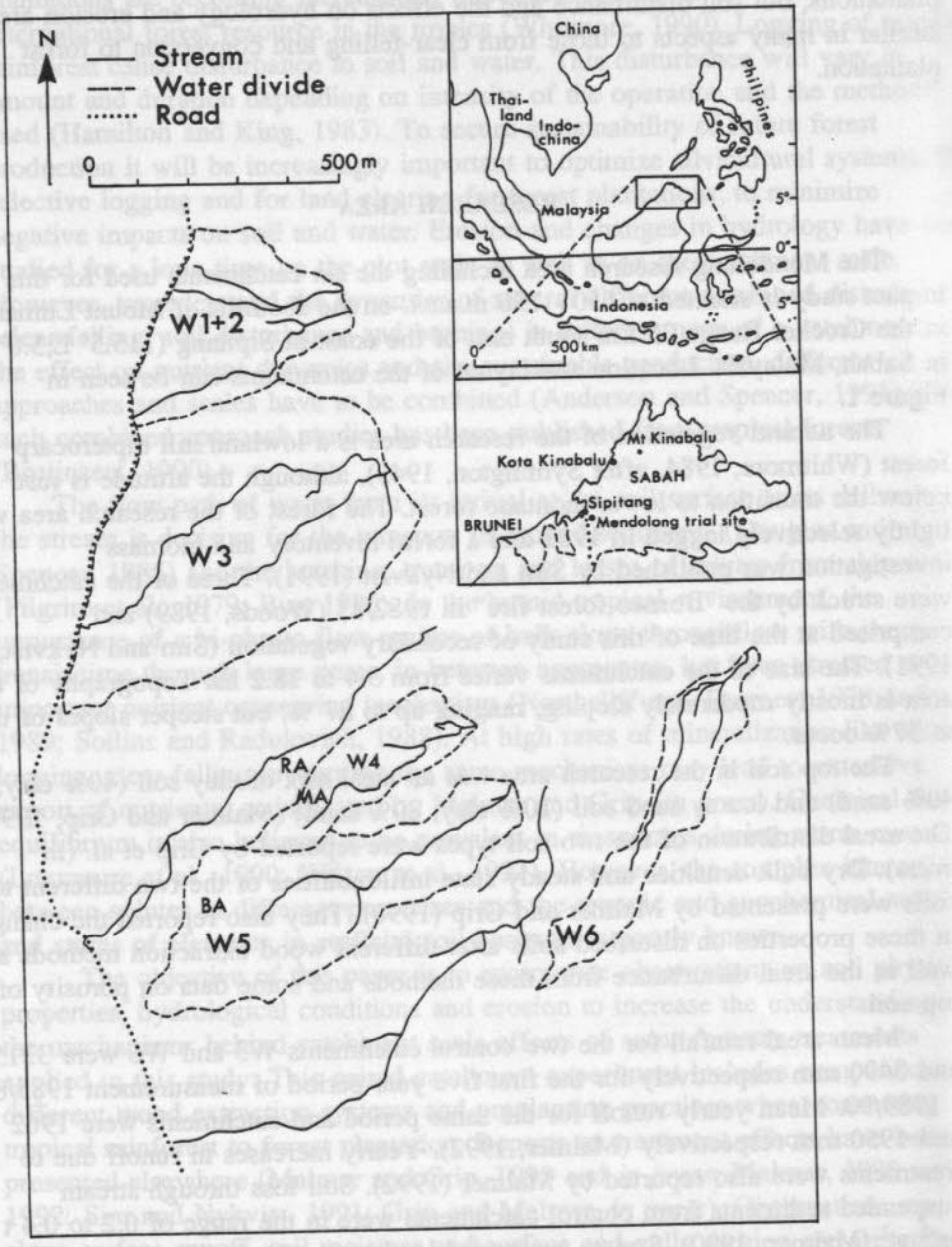

Figure 1. Location and layout of the catchments in Mendolong research area, Sabah, Malaysia. RA, MA and BA denote the location of the main runoff plots used in this study.

METHODS

Forestry treatments

The treatments, covering 100 % of the catchments, were as follows:

W1 Non-mechanized clearing of remaining trees and secondary vegetation, no wood extraction, burning of all biomass and planting.

W2 The same treatment as W1.

W3 Control catchment for forest fire area. No treatments.

W4 Manual felling, manual wood extraction, clearing of planting rows in the slash and planting in these rows, without burning.

W5 Manual felling, wood extraction using crawler tractors, burning of the remaining biomass and planting (normal practice).

W6 Control catchment for selectively logged area. No treatments.

Felling and extraction was carried out from November 1987 to January 1988. Burning and rowing was done in February to April, and planting was conducted at W1+2 and W4 in March and W5 in May 1988. Wood extraction methods used were described in detail by Malmer and Grip (1990). All logging and silvicultural treatments in this study were preformed by one of the regular contractors and personnel involved in full scale operations in the surrounding areas.

The normal practice of burning the residual biomass before planting was used in W1+2 and W5. About one month after burning, grasses and weeds started to colonize the bare soil, and after three months most soil, except for tractor disturbed areas, was covered with vegetation. At W4 burning was avoided and all remaining biomass was cut into smaller constituents and put into rows, leaving strips of bare ground accessible for planting. The planted tree species was *Acacia mangium* for all treated catchments. The accumulation of above ground biomass in trees and ground vegetation in the plantations after 1.5 years were in W1+2: 2.3, 6.2; W4: 10.5, 1.8 and W5: 5.4, 3.3 t ha^{-1} respectively (Sim and Nykvist, 1991).

Field measurements

Except from the network of 12 raingauges with weekly recording in the research area, 4 raingauges adjacent to the runoff plots were recorded 6 days per week. Hourly time and intensity of precipitation was recorded by use of a OTA tipping bucket recorder with weekly charts. Observations of soil moisture were made by use of tensiometers installed in profiles at 20 and 50 cm depth in the highest and the lowest points of runoff plots **RA** (Reference), **MA** (Manual and no burning, W4) and **BA** (Burning, W5) (Table I and below). Readings were taken once a day, six days a week, between 7 - 9 am. Soil moisture recordings where only point measurements, and did not include any replicates.

Large unbounded runoff plots were installed only on clay top soil (cf. above) in reference forest (**RA**) in December 1987, in W4 after manual wood extraction (**MA**) in February 1988 and in W5 after burning (**BA**) in March 1988. The plots in the clear-felled catchments W4 and W5 were located on soil undisturbed by wood extraction. Number of plots and their area, length and slope are presented in Table I.

Plots RA, RB, RC, MA, MB and MC were all placed together on each side of the northwest waterdivide of W4, which consequently was the upper boundary of all these plots (Figure 1). Plot BA was located in the upper part of W5, in 300 m distance from the other plots. Plots RB, RC, MB and MC were adjacent to RA and MA respectively, but were collecting water from smaller parts of the slope, starting from the water divide. All plots were installed on topographically similar slopes of convexo-concave form. The lengt of the main plots RA, MA and BA were not the same as homogenity in soil type and proximity were preferred. Also the length of the plots were characteristic of the slopes of the respective catchments. On the other hand the area of these three plots were similar, meaning that the longer slopes were more narrow in the up slope part. As expected, surface erosion and runoff were depending on lenght of slope, as indicated by the results from the partial slope plots. However, this bias was considered to be compensated by the similar area of main plots, and comparisons between control and treatments would not over-estimate the effects of treatments.

Water and sediment was collected in dug down galvanized sheet-metal gutters of 7 m length. The collecting gutter was attached to the soil in the upper surface of the mineral soil and thereby collecting all water running off in the litter and the thin humus layer, or in the ash and remaining humus in BA. The drainage area for each plot was determined from the surface water divide for the mini catchment of the collecting gutter. Collected water volumes were measured by carefully calibrated tipping buckets of about 5 l volume. Volumes counted by the tipping buckets were recorded in the morning 6 days per week. Surface runoff detection limits were 0.05 mm day^{-1} for main slope plots and

Table I. Description of unbounded runoff plots in Mendolong research area in Sabah, Malaysia. All plots were located on soils undisturbed by wood extraction. ***R****-plots denote reference forest,* ***M****-plots manual extraction and no burning after clear-felling (W4) and* ***B****-plots burning after clear-felling (W5).*

plot	Area (m^2)	Length (m)	Slope % mean	Slope % interval	Treatment
RA	192	40	27.7	22 - 50	Reference forest, main plot
RB	108	17	36.4	22 - 45	like RA, upper half of slope
RC	55	7	19.6	18 - 23	like RA, top of slope
MA	198	33	37.4	23 - 54	Manual extraction, no burning (W4)
MB	111	18	26.9	5 - 56	like MA, upper half of slope
MC	61	7	42.8	15 - 51	like MA, top of slope
BA	194	25	31.7	17 - 44	Burning after clear-felling (W5)

0.1 mm day^{-1} for the smaller partial slope plots. A slot funnel was collecting a calibrated subsample of about 10 % of the water volume to large storage drums. The amount of sediments in the stored water was determined by filtration. The detection limit for eroded sediments from the runoff plots was estimated to 1 kg ha^{-1} day^{-1}. To make room for measuring and storing devices, the collecting gutters of the plots RA and MA were placed about 10 m from the very stream channel in the gently sloping concave bottom of the slope. That means that these plots only measured the surface runoff in the very slopes, and did not detect any possible saturation overland flow or return flow (Dunne, 1978) adjacent to the stream. Plot BA was placed in a slope similar to MA in form, but which did not have a stream channel in the bottom. This was because there were no full length slope adjacent to the stream in W5 with only clay top soil.

In 1991 a small inventory of soil surface animal burrows was conducted in the RA plot. The inventory was made on 5 m^2 in the lower part of the slope. The litter layer and the thin humus layer was carefully removed and burrows into the mineral soil was counted in 2 diameter classes. Afterwards water was carefully pored into some of the burrows to investigate the rate of water percolation through such macropores.

Saturated hydraulic conductivity was measured in the field using the inversed auger hole method (Kessler and Oosterbaan, 1974) slightly modified by Messing (1989) and Messing and Jarvis, (1990). Measurements were taken in December 1989 at 20 and 40 cm depth on clay and loamy sand on tractor tracks in W5 and in surrounding forest (control) (Andersson, 1990). Ten replicates were made on every soil and treatment category totalling 80 measurements.

Gully formation on tractor tracks was recorded in December 1989 and 1990. This was made using quickset level to determine the change in soil surface at permanent profiles transverse to tractor tracks. Three stretches of main tractor tracks were chosen to reflect the worst case of point erosion inside the catchment to put in relation to the sediment delivery recorded in the stream. The measured transects were 10 m apart and total length of slopes 45 - 70 m. Soil loss from these slopes was estimated by multiplying lost volumes calculated from surface measurements and available means of soil bulk density (Malmer and Grip, 1990).

Data analysis

Data used here are from January - December 1988. Autocorrelations of daily and maximum hourly rainfall and daily surface runoff were non significant. Relations between maximum hourly rain intensity, 24 hour rain and plot runoff per event were tested by regression analysis on 10-log transformed data.

Differences between means of saturated hydraulic conductivities of different soil and treatment categories were made using a non parametric multiple test (Dunn, 1964; Andersson, 1990).

RESULTS

Rain and hydraulic conductivity

The year of measurements reported here, 1988, was the most humid recorded at the experimental area for 5 years (Malmer, 1992). Total rainfall at the plots for 1988 was 4352 mm. Distributions of 24 hour rainfall and maximum hourly rainfall per event are presented in Figure 2. Number of days with recordings of rain 1988 was 190, but 55 of these had less than 8 mm of rain. More than 50 mm of rain was recorded during 21 days and the maximum rainfall during 24 hours was 111 mm. Maximum hourly rain intensity was less than 30 mm, but higher intensities of 5 - 10 minutes duration were noticed.

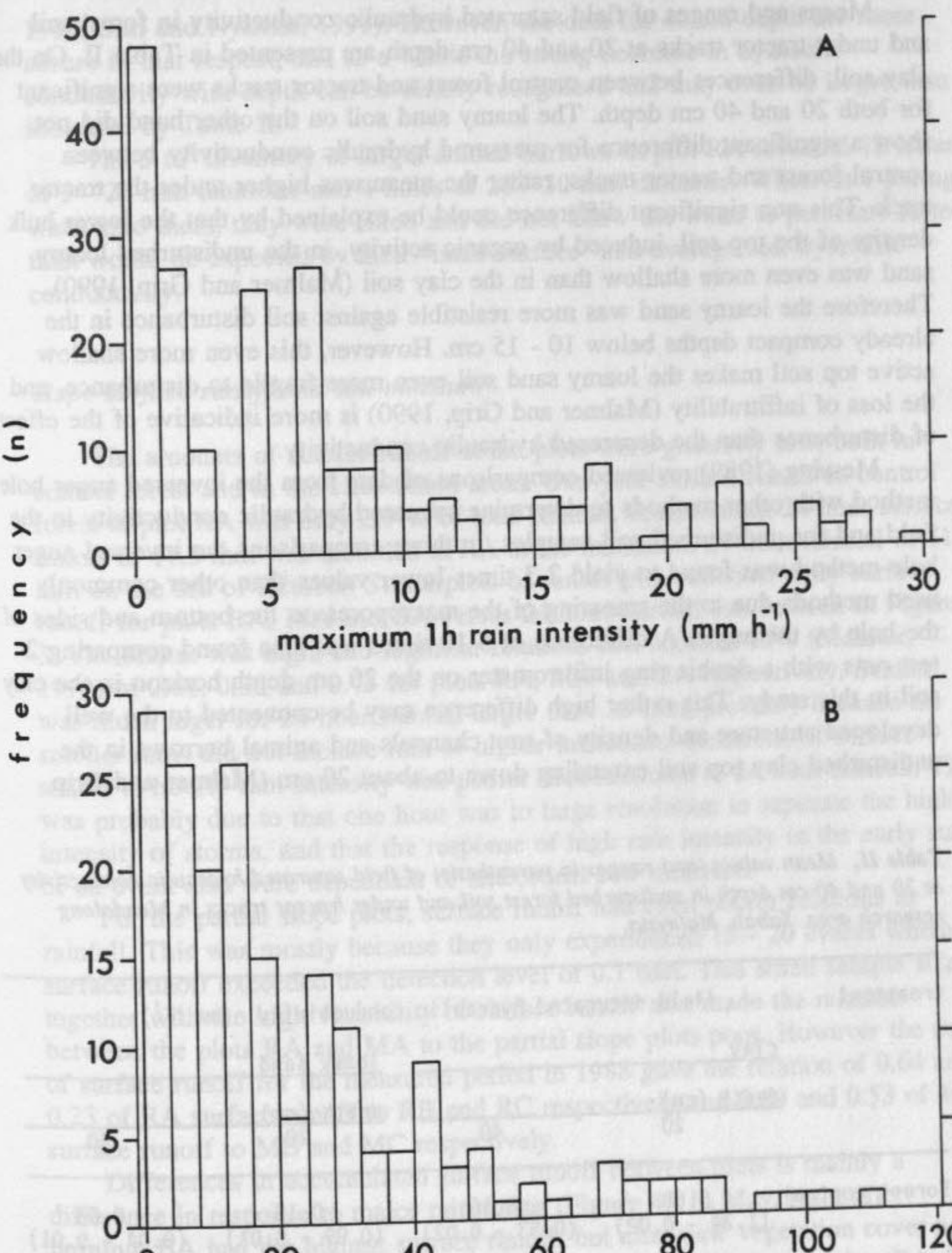

Figure 2. Distributions of maximum 1 hour rain intensity per rain event (A) and 24 hour rainfall (B) at runoff plots in Mendolong research area for the year of 1988.

Means and ranges of field saturated hydraulic conductivity in forest soil and under tractor tracks at 20 and 40 cm depth are presented in Table II. On the clay soil, differences between control forest and tractor tracks were significant for both 20 and 40 cm depth. The loamy sand soil on the other hand did not show a significant difference for measured hydraulic conductivity between control forest and tractor tracks, rather the mean was higher under the tractor track. This non significant difference could be explained by that the lower bulk density of the top soil, induced by organic activity, in the undisturbed loamy sand was even more shallow than in the clay soil (Malmer and Grip, 1990). Therefore the loamy sand was more resistible against soil disturbance in the already compact depths below 10 - 15 cm. However, this even more shallow active top soil makes the loamy sand soil even more fragile to disturbance, and the loss of infiltrability (Malmer and Grip, 1990) is more indicative of the effect of disturbance than the decreased hydraulic conductivity.

Messing (1989) reviewed comparisons of data from the inversed auger hole method with other methods to determine saturated hydraulic conductivity in the field and for undisturbed soil samples. In those comparisons the inversed auger hole method was found to yield 2-3 times lower values than other commonly used methods due to the smearing of the macropores on the bottom and sides of the hole by the auger. A difference of 5-10 times was also found comparing 2 test runs with a double ring infiltrometer on the 20 cm depth horizon in the clay soil in this study. This rather high difference may be connected to the well developed structure and density of root channels and animal burrows in the undisturbed clay top soil extending down to about 20 cm (Malmer and Grip,

Table II. ***Mean values (and ranges in parenthesis) of field saturated hydraulic conductivity at 20 and 40 cm depth in undisturbed forest soil and under tractor tracks in Mendolong research area, Sabah, Malaysia.***

treatment	field saturated hydraulic conductivity ($mm\ h^{-1}$)			
	clay		loamy sand	
	depth (cm) 20	40	depth (cm) 20	40
Forest control	0.68 (1.34 - 0.02)	0.16 (0.53 - 0.02)	0.10 (0.09 - 0.01)	0.03 (0.04 - 0.01)
Tractor track	0.03 (0.16 - 0.06)	0.01 (0.06 - 0.01)	0.18 (0.53 - 0.06)	0.05 (0.10 - 0.02)

1990; Sim and Nykvist, 1991). However, the data for 40 cm depth are more secure in that respect, and as a whole the strong decrease in hydraulic conductivity with depth can be clearly recognized and may even be larger than indicated by Table II.

The 5 m^2 inventory of larger animal burrows in plot RA revealed 12 holes of 5 - 20 mm diameter and 4 holes of 20 - 50 mm diameter. When later poring water into them, they were filled and did not allow the water to percolate faster than would be expected by their "inner surface" and average soil hydraulic conductivity.

Slope surface runoff and soil moisture

The amounts of surface runoff at the plots were generally low, both in control forest and in the clear-felled areas. One year surface runoff in control forest at plot RA was only 2.9 % of total rainfall. A maximum 24 hour surface runoff of 17.5 mm was recorded at RA at the maximum 24 hour rainfall of 111 mm on the 3rd of October. Scatterplots of runoff plot collected daily surface runoff for plots RA, MA and BA versus 24 hour rainfall can be seen in Figure 3. The scatter was high, and loglinear relations had adjusted R^2's (Malmer, 1992) of 0.61, 0.36 and 0.43 for plots RA, MA and BA respectively. Scatter was much higer for 24 hour rainfall larger than 30 mm, probably because the smaller rains did not include rain of higher intensities. Relations of surface runoff to hourly rain intensity was poorer than relations to 24 hour rainfall. This was probably due to that one hour was to large resolution to separate the highest intensity of storms, and that the response of high rain intensity in the early stage of an event also were dependant of antecedent soil moisture.

For the partial slope plots, surface runoff had even poorer relations to rainfall. This was mostly because they only experienced 15 - 20 events where surface runoff exceeded the detection level of 0.1 mm. This small sample size together with the high variability of surface runoff also made the relation between the plots RA and MA to the partial slope plots poor. However the sums of surface runoff for the measured period in 1988 gave the relation of 0.64 and 0.23 of RA surface runoff to RB and RC respectively and 0.69 and 0.53 of MA surface runoff to MB and MC respectively.

Differences in accumulated surface runoff between plots is mainly a difference in response to major rainstorms (Figure 4). In May, right after burning, BA had the highest surface runoff, but after new vegetation cover had established the response to rainstorms was more moderate compared to RA. In the most rainy period of the year, October - November, RA showed the highest response to major rain events, especially the one on the 3rd of October mentioned above. During the same time MA experienced the smallest responses in surface runoff to major rain events, and plot BA held an intermediate position. That was also in line with the observations of the response of soil suction to the treatments.

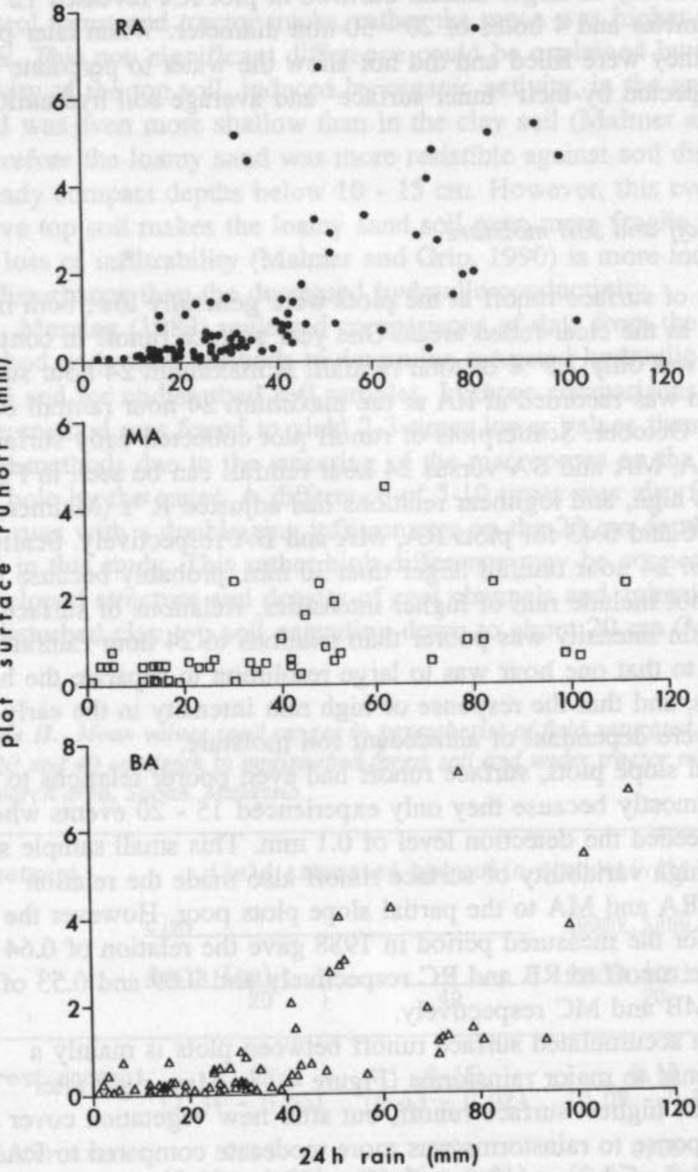

Figure 3. Scatterplot of surface runoff versus 24 hour rainfall at Mendolong research area, Sabah, Malaysia in 1988. RA - Reference forest, MA - Manual extraction, no burning and BA - Burning after clear-felling.

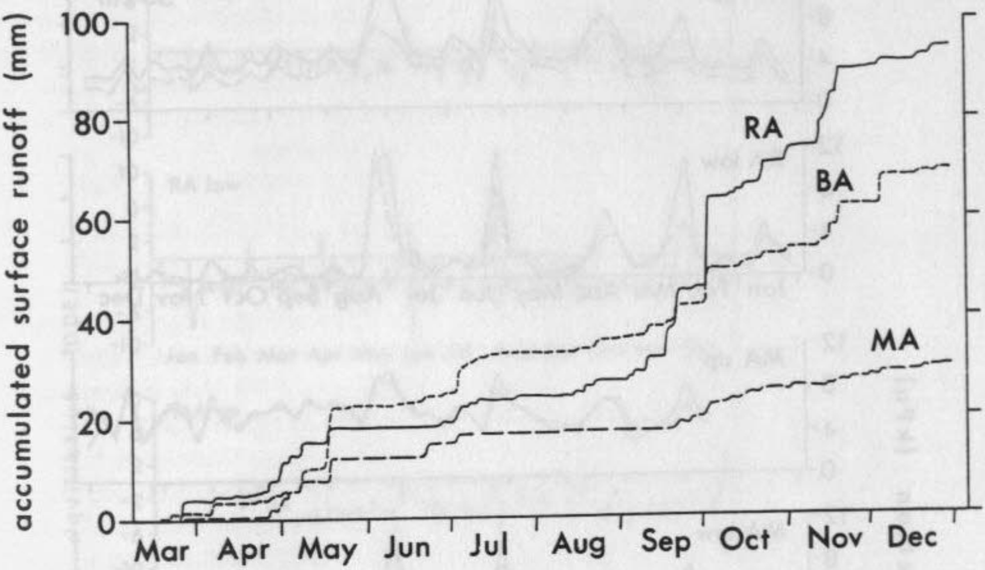

Figure 4. *Accumulated surface runoff at Mendolong research area, Sabah, Malaysia during 9 months 1988.* ***RA*** *- Reference forest,* ***MA*** *- Manual extraction, no burning and* ***BA*** *- Burning after clear-felling.*

The clay top soils of the research area were generally wet. Measured soil suction at 20 cm depth were seldom above 20 kPa under forest cover. Only after weeks with low rainfall and clear days, such suctions could be noticed. Figure 5 shows weekly means of measured soil suctions at 20 and 50 cm depth in the top and the bottom of the plots RA, MA and BA (Table I). The lowest part of the plots were wetter than the upper parts of the slopes. Note that soil horizons at 20 cm generally had lower suctions than drainage equilibrium compared to suctions at 50 cm (i.e. the soil suction at 20 cm was less than 3 kPa larger than 30 cm below in the 50 cm horizon). In Figure 6 the daily "deviation from drainage equilibrium" (DDE) is shown for the recordings of soil suction reported in Figure 5. DDE is defined as:

$$(\text{suction at 20 cm} - \text{suction at 50 cm}) - 3 \quad (\text{kPa})$$

A negative value of DDE means that drainage occurred from the 20 cm horizon. The only exception from the general pattern was observed at the water divide position of plot BA in the burned area, as the top soil experienced more evaporation without a shading vegetation cover and the top slope position could not recieve any soil water from the sides. However, as vegetation recovered in the later part of the year, there was a tendency of that relation to approach a

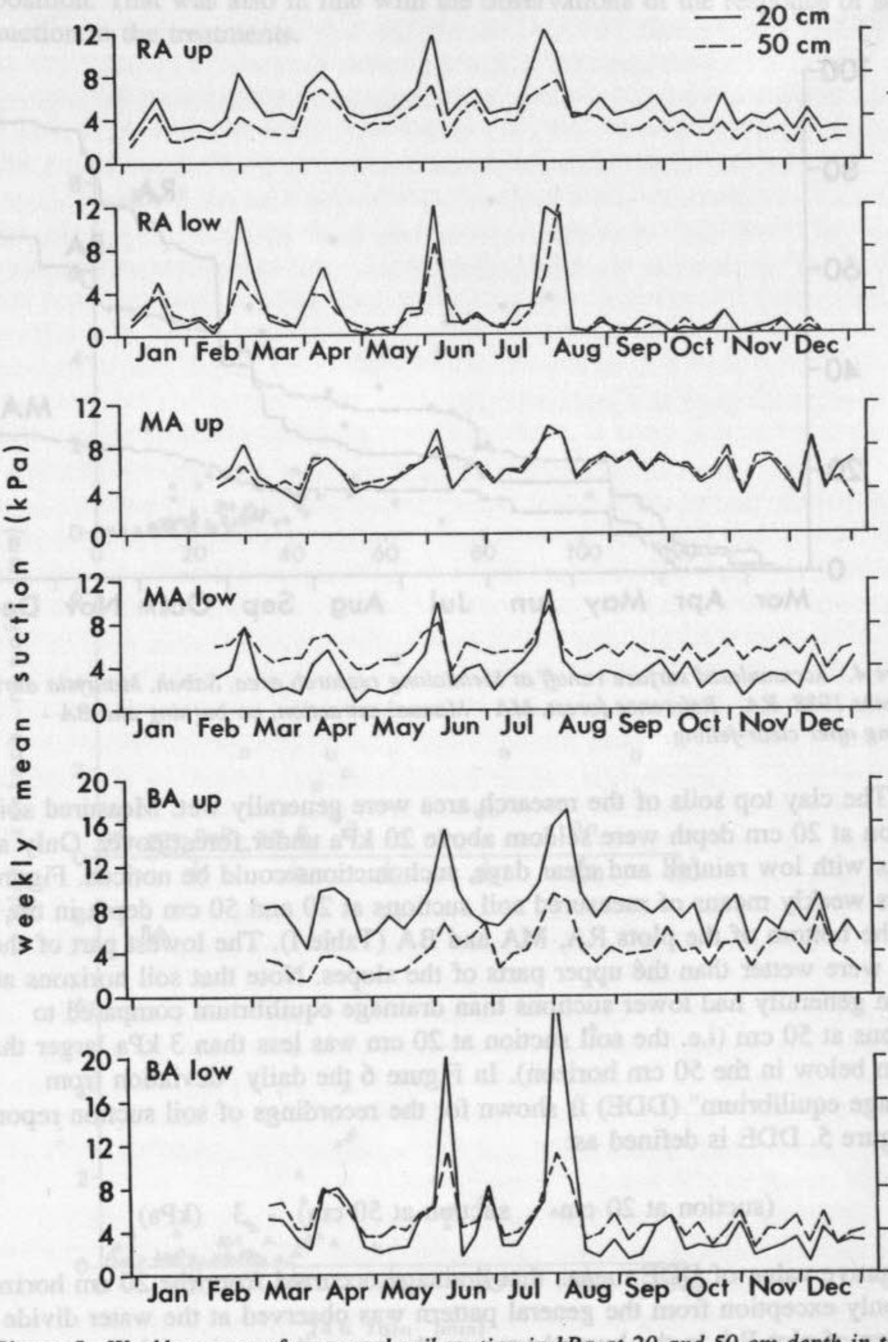

Figure 5. *Weekly means of measured soil suction in kPa at 20 and 50 cm depth in the highest and lowest points of runoff plots in 1988 in Mendolong research area, Sabah, Malaysia. RA - Reference forest, MA - Manual extracton, no burning and BA - Burning after clear-felling.*

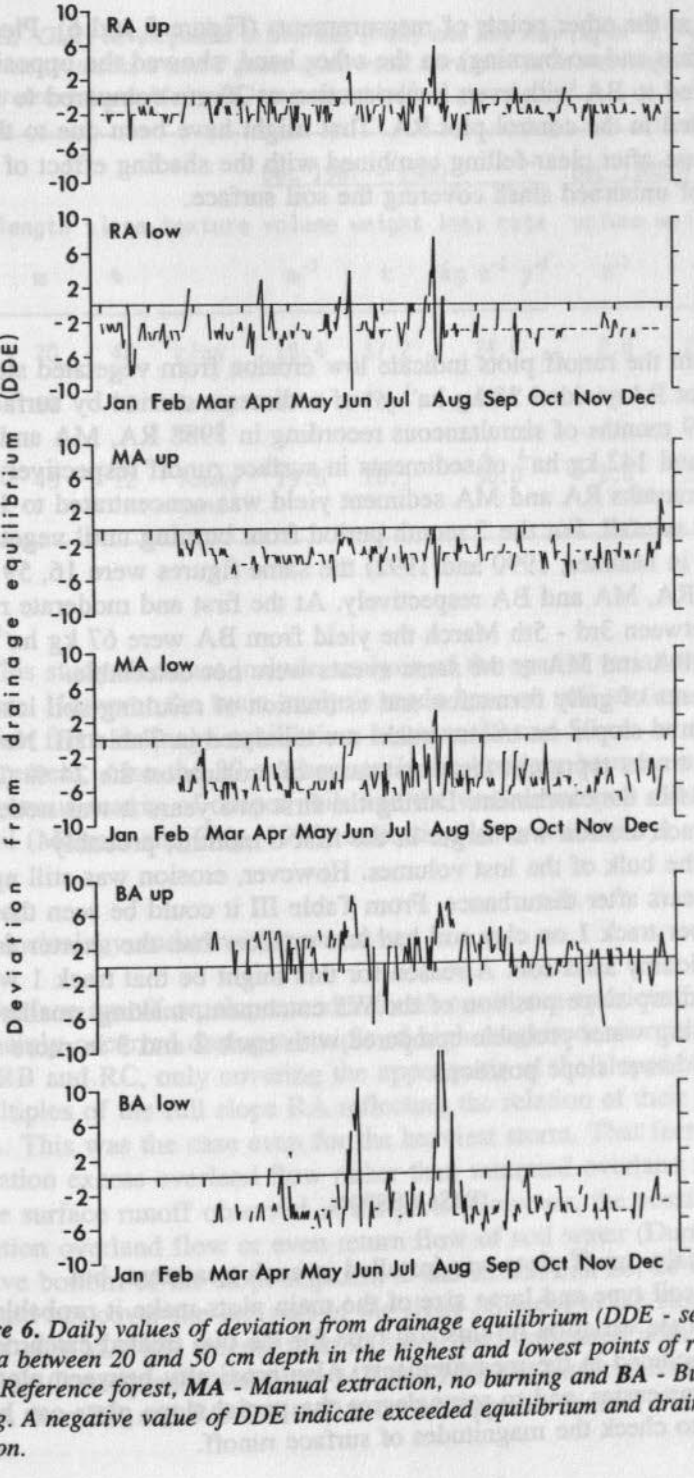

***Figure 6.** Daily values of deviation from drainage equilibrium (DDE - se text for explanation) in kPa between 20 and 50 cm depth in the highest and lowest points of runoff plots in 1988. RA - Reference forest, MA - Manual extraction, no burning and BA - Burning after clear-felling. A negative value of DDE indicate exceeded equilibrium and drainage from the 20 cm horizon.*

similar state as at the other points of measurements (Figure 5 and 6). Plot MA (manual extraction and no burning) on the other hand, showed the opposite relation compared to BA with even lower suction at 20 cm compared to 50 cm than was recorded in the control plot RA. That might have been due to the lowered water use after clear-felling combined with the shading effect of the large amounts of unburned slash covering the soil surface.

Erosion

Results from the runoff plots indicate low erosion from vegetated slopes. In 1988 control plot RA yielded 38 kg ha^{-1} y^{-1} of sediments carried by surface runoff. For the 9 months of simultaneous recording in 1988 RA, MA and BA yielded 27, 82 and 142 kg ha^{-1} of sediments in surface runoff respectively. During these 9 months RA and MA sediment yield was concentrated to 14 events of heavy rainfall. For the 5 month period from burning until vegetation cover (period 3 in Malmer, 1990 and 1992) the same figures were 16, 59 and 136 kg ha^{-1} for RA, MA and BA respectively. At the first and moderate rains after burning between 3rd - 5th March the yield from BA were 67 kg ha^{-1}, while the yields from RA and MA at the same events were not detectable.

Measurements of gully formation and estimation of resulting soil loss from the three monitored slopes on tractor tracks are tabulated in Table III. Note that the investigated tracks represent the worst cases of erosion on the 24 % (2.3 ha) of disturbed soils in the catchment. During the first two years it was noticed that rate of tractor track erosion was larger in the first 6 months, probably contributing to the bulk of the lost volumes. However, erosion was still apparent in 1990, three years after disturbance. From Table III it could be seen that the longer and steeper track 1 on clay soil had less erosion than the shorter and less steep tracks on loamy sand soil. A reason for this might be that track 1 was in the upstream and top slope position of the W5 catchment, making smaller volumes of eroding water probable compared with track 2 and 3 in more downstream and lower slope positions.

DISCUSSION

Only one main runoff plots was installed in each treatment, but homogeneity in soil type and large size of the main plots make it probable that apparent small scale variation on this soil type for the two treated catchments discussed was included in the measurements. Also proximity between plots make comparisons easier, and to some degree the partial slope plots can be seen as replicates as to check the magnitudes of surface runoff.

Table III. Gully development in soil loss (t dw) and loss rate (kg m^{-2} y^{-1}) on 3 severely eroded tractor tracks 2 and 3 years after wood extraction in Mendolong, Sabah, Malaysia. nd = not determined. Multyply loss rate with 10 to get t ha^{-1} y^{-1}.

				Dec 1987 - 1989			Dec 1989 - 1990		
track	length	slope	texture	volume	weight	loss rate	volume	weight	loss rate
	m	%		m^{-3}	t	kg m^{-2} y^{-1}	m^{-3}	t	kg m^{-2} y^{-1}
1	70	48	clay	16.4	17.2	24.4	2.6	2.7	7.7
2	60	18	loamy sand	17.6	26.7	54.7	nd	nd	nd
3	45	12	loamy sand	19.5	20.5	50.0	1.6	2.5	12.2

This study does not include analysis of the spatial variation of soil moisture. However the main analysis made here on soil suction concerns deviation from drainage equilibrium between 20 and 50 cm in each profile of measurement. Also the 20 cm horizon is the boudary between a fine textured homogenous matrix of deeper soils and the well developed soil structure of the top soil (Malmer and Grip, 1990) indicating lower variability.

Slope hydrology under rainforest cover

Surface runoff on slopes under forest cover was restricted to minor volumes and mainly occurred during a couple of high magnitude storms in 1988. The plots RB and RC, only covering the upper parts of the slopes had surface runoff in multiples of the full slope RA reflecting the relation of their length compared to RA. This was the case even for the heaviest storm. That fact speaks for infiltration excess overland flow rather than saturated overland flow as reason for the surface runoff observed in the plots. However, the possibility of saturation overland flow or even return flow of soil water (Dunne, 1978) on the concave bottom of the slope adjacent to the stream can not be excluded as the plots did not cover the last part of the slope adjacent to the stream. The upper part of plot MA was steeper than its lower part compared with plot RA (cf. Table I). Consequently the plots MB and MC had larger surface runoff multiples of MA than RB and RC had to RA on the opposite side of the same water divide.

The soil bulk density successively increasing with depth in the upper 20 cm indicate a gradient of saturated hydraulic conductivity between 154 mm h^{-1} in the uppermost horizon at the surface of the mineral soil (Malmer and Grip, 1990) and 0.68 mm h^{-1} at 20 cm. Also, as indicated above, this later hydraulic conductivity presented for 20 cm in the forest clay top soil might be suffering from a larger underestimation than from the other horizons measured. A further decrease in hydraulic conductivity below 40 cm could be expected as Malmer and Grip (1990) reported a higher clay content of 55 % below 45 cm compared to 40 % above. Using a formula from Campbell (1985):

$$K_s = C \exp(-6.9 m_c - 3.7 m_s) \quad (\mathrm{kg\ s\ m^{-3}})$$

a saturated hydraulic conductivity of 0.047 mm h^{-1} at 50 cm depth was calculated from textural data in (Malmer and Grip, 1990). K_s is saturated hydraulic conductivity and m_c and m_s denote clay and silt mass fractions respectively. Campbell (1985) recomend a value of 4 * 10^{-3} for the constant C. Here it was given the value 1.77 * 10^{-3} by substitution of K_s, m_c and m_s for the field measured data for the 40 cm horizon.

The high magnitude of rainfall, few and short dry periods (Malmer, 1992) and consistent low soil suction in top soil paired with no slope surface runoff and rapidly decreasing saturated soil hydraulic conductivity with depth, speaks for shallow troughflow (Kirkby and Chorley, 1967) as the main transport mechanism of water to the stream. The occurrence of sub surface stormflows (Beven and Germann, 1982) in large macropores induced by animal burrows and rot channels may occur to some extent in the research area. At excavation of soil profiles, observations of higher rates of water flowing along roots were common. On the other hand the small inventory of larger animal burrows did not indicate them to be of major importance. Rather, very low bulk density in the top 10 cm of the soil (Malmer and Grip, 1990) speaks for the importance of smaller organisms to maintain aggregated soil structure and meso and macro porosity in and between aggregates. With the high top soil root density (Sim and Nykvist, 1991) any tendencies of sub surface stormflows would have been included in the variation of the extensive measurements of infiltration and hydraulic conductivity.

Concequently, as macropore subsurface stormflow was not considered to be of major importance for stream runoff generation, the fast response of the streamflow to rainfall (Malmer, 1992) must be attributed to translatory flow of shallow soil water close to the stream being "pushed out" by displacement as new water entered the soil pores on the slope surface (Hewlett and Hibbert, 1967; Bonell et al., 1990). Sklash and Favolden (1979) introduced the groundwater ridging theory to explain the fast response of pre-event water in the rise of streamflow hydrographs. By a fast saturation of the capillary fringe (horizons close to saturation above the groundwater table) nearby the stream, the groundwater table rises most rapidly in the wetter lower part of the slope. As a result the hydraulic gradient towards the stream increases and triggers groundwater outflow. In the situation described here, a faster rise of the groundwater level in the lowest part of the slope seems likely to explain the fast response of stream stormflow to rainfall. As stated above extensive surface runoff were not apparent in the very slopes. However, saturated overland flow or returnflow (Dunne, 1978) in the flatter area adjacent to the stream, not included by the plots, could not be excluded. Also the lower soil suction in the lowest point of plot RA indicate that more rapid saturation of top soil adjacent to the stream were likely, which also can be supported by theoretical calculation of the flow of soil water under the collecting gutter of plot RA:

In Table IV saturated hydraulic conductivity for different depth intervals down to 2.5 meters are tabulated. These were derived by interpolation of the logarithm of the conductivities of 20, 40 and 50 cm together with steady state infiltrability. The conductivity below 50 cm was assumed to equal that of the 50 cm horizon. From data on slope in Table I the lateral gradient of the lower part of plot RA can be concluded to be 0.22 m m^{-1}. The maximum flow through a 2.5 m deep cross section below the 7 m collecting gutter of the plot was calculated from Darcy's law with Dupuits assumptions of horizontal flow. The maximum gradient was equal to that of the soil surface.

***Table IV**. Interpolated saturated hydraulic conductivity, K_s (mm h^{-1}) of different depth intervals and calculated maximum soil water flow (l h^{-1}) through a 2.5 m deep and 7 m wide cross section below the base of runoff plot RA.*

depth interval (m)	K_s (mm h^{-1})	maximum soil water flow (l h^{-1}) per horizon	accumulated
0.5 - 2.5	0.047	0.14	0.14
0.4 - 0.5	0.1	0.014	0.154
0.2 - 0.4	0.42	0.126	0.280
0.15 - 0.2	0.84	0.049	0.343
0.10 - 0.15	1.86	0.140	0.483
0.05 - 0.10	7.46	0.574	1.057
0 - 0.15	85.1	6.552	7.609

As can be seen from Table IV only 7.6 l h^{-1}, equivalent to 0.04 mm on the plot surface, could flow through a 2.5 m deep cross section below the plot. Concequently, stream stormflow peaks must to a high degree be supplied by saturated overland flow and returnflow close to the stream.

Figure 6 may be interpreted as follows: The almost constantly lower soil suction than drainage equilibrium at 20 cm compared with 50 cm means that drainage was apparent at almost all time at 20 cm (except for top of slope in plot BA). However vertical drainage was likely to be very slow because of the drastically lowered hydraulic conductivity with depth. That means that soil moisture and total potential was built up in the top soil until structural pores sufficiently large for lateral drainage of the well aggregated top soil were activated. That level is indicated in Figure 6 by the flat level visual in the lower part of the graphs. The appearance of a critical level is obvious in the graph, but it was not absolute over time as the soil suction at 50 cm was not constant.

The different critical levels of deviation from drainage equilibrium (DDE) at different points of recording could indicate small differences in the pore size distribution and soil structure in the top soil. A tendency of a lower critical level can be seen for the down slope points in all plots. That could be likely as longer times of saturation than in top of the slopes would mean worse conditions for soil aggregation.

Peaks in the graphs above the critical level of lateral drainage, and especially those of longer duration due to dry periods, were often followed by a lower DDE than the critical level for lateral drainage. Such situations could be interpreted as cases when both 20 and 50 cm horizons had experienced raised soil suction and rain had wetted the top soil, but slow vertical drainage had not yet reduced soil suction in the lower horizon as much, giving a low value of DDE.

Figure 7 is a scatterplot of mean stream baseflow in control stream W3 (control catchment on 100 % clay soil) between 6 and 10 am and the corresponding DDE for recordings of soil suction in the lowest point of plot RA. Figure 7 also indicate actual soil suction at 20 and 50 cm for every measurement by type of symbol (20 cm) and size of symbol (50 cm). The points above zero in DDE indicate the dry periods when low lateral or vertical drainage in top soil were apparent and only low stream baseflow with water draining from deeper horizons with lower permeability occurred. Consequently this was also the driest situations for both horizons.

On the other hand the points with the lowest DDE represent days after dry periods when the 50 cm layer is not yet rewetted to the same degree as for the 20 cm horizon (cf. above). In that situation baseflow was still low and dominated by water from deeper horizons.

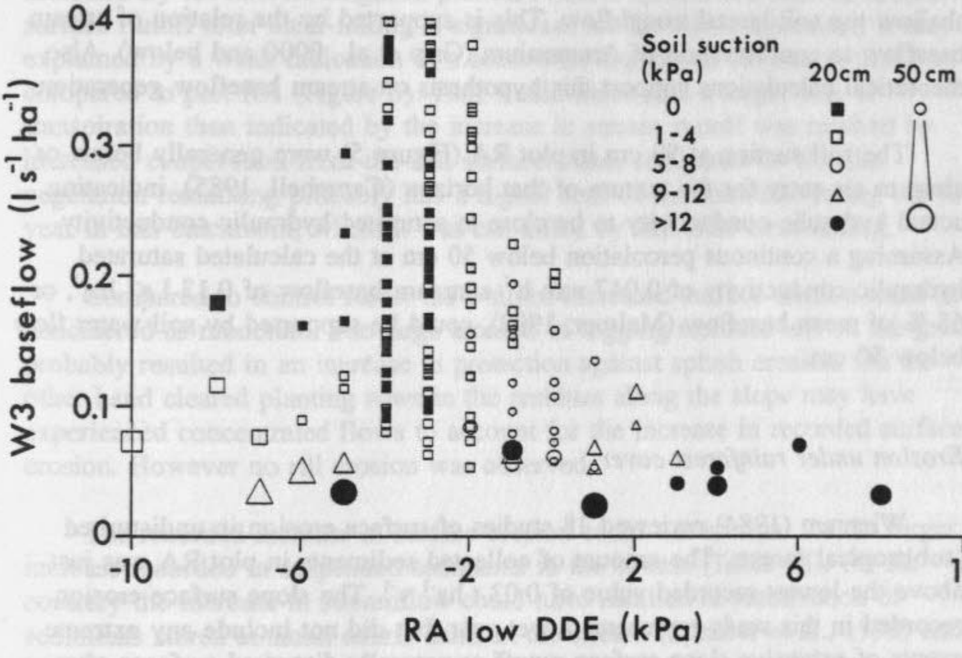

Figure 7. Scatterplot of mean specific stream baseflow (6 - 10 am) for control catchment W3 stream versus deviation from drainage equilibrium (recorded at 7 - 9 am) for lowest point of the slope in control runoff plot RA. Actual soil suction at 20 and 50 cm for every measurement is marked by type of symbol (20 cm) and size of symbol (50 cm).

High baseflows, above mean stream baseflow, 0.2 $l\ s^{-1}\ ha^{-1}$ (Malmer, 1992), only occurred around the "critical level for lateral drainage" in top soil (cf. Figure 6). These were also the situations with the lowest suctions in both horizons. However, at that level of DDE all baseflows except for the very lowest occurred. That high variation may be explained by a variety of situations in between the circumstances for the two situations described above. It should also be noted that plot RA were not draining to the W3 stream (Figure 1), but to another stream in 300 m distance not used for this study. However, W3 were chosen because it was the control catchment of 100 % the same soil type as plot RA. Despite the different catchment, the agreement in Figure 7 must be considered good.

A conclusion from the discussion from Figure 6 and 7 above may be that not only stream stormflow but also higher stream baseflows were dependant on shallow top soil lateral troughflow. This is supported by the relation of stream baseflow to concentration of Ammonium (Grip et al., 0000 and below). Also theoretical calculations support this hypothesis on stream baseflow generation:

The soil suction at 50 cm in plot RA (Figure 5) were generally below or close to air entry for the texture of that horizon (Campbell, 1985), indicating actual hydraulic conductivity to be close to saturated hydraulic conductivity. Assuming a continous percolation below 50 cm at the calculated saturated hydraulic conductivity of 0.047 mm h^{-1} a stream baseflow of 0.13 l s^{-1} ha^{-1}, or 65 % of mean baseflow (Malmer, 1992), could be supported by soil water flow below 50 cm.

Erosion under rainforest cover

Wiersum (1984) reviewed 18 studies of surface erosion in undisturbed (sub)tropical forests. The amount of collected sediments in plot RA was just above the lowest recorded value of 0.03 t ha^{-1} y^{-1}. The slope surface erosion recorded in this study represents a wet year, but did not include any extreme events of extensive slope surface runoff or naturally disturbed surfaces after treefalls or small landslides. Consequently, the surface wash apparent was not enough to reactivate sediments stored in small sediment traps by roots and larger debris along the slope. Traces of splash erosion was observed in plot RA and other slopes under forest cover, but no erosion by surface wash or rill were observed on undisturbed slope surfaces.

Effects of clear-felling without subsequent burning

Surface runoff collected from plot MA, in catchment W4 with clear-felling and no subsequent burning, was even smaller than from plot RA in control forest. In this catchment the increase in stream runoff after clear-felling could not significantly be related to stream base- or stormflows (Malmer, 1992). The later fact was concluded to relate to that natural water ways were not disturbed in this catchment. The observations presented in this article supports that conclusion. Slope surface runoff did not increase, despite decreased forest interception and transpiration. This indicates that the mechanism of high steady state infiltrability and shallow troughflow still was "capable of managing" the increased volumes of water without resulting in overland flows.

The surface runoff in plot MA compared to plot RA was even smaller than indicated by the shorter length in plot MA (Table I). If this observed decrease of surface runoff after clear-felling is significant for the whole operation, it may be explained by a weak indication of a somewhat higher soil suction in plot MA compared to plot RA (Figure 5). That would mean that a larger loss of transpiration than indicated by the increase in stream runoff was masked by increased evaporation from the soil surface. Also, compared to W5, the vegetation remaining probably had a higher sum of transpiration during the first year in this catchment, where it was not killed by fire after clear-felling.

Compared to control forest the 3 times increased surface erosion must be concidered as moderate. The large amount of logging residues left on the ground probably resulted in an increase in protection against splash erosion. On the other hand cleared planting rows in the residues along the slope may have experienced concentrated flows to account for the increase in recorded surface erosion. However no rill erosion was observed.

The recorded increase in surface erosion could not account for the larger increase recorded in suspended sediments in the stream (Table III). On the contrary the increase in streamflow could have resulted in reactivation of sediments stored at small debris dams in the stream (Spencer et al., 1990) and by scour and channel widening at logging residues fallen into the stream (cf. Malmer, 1990). Bons (1990) also showed delayed extensive flushing of trapped sediments from clearing of the stream channel.

Effect of clear-felling and subsequent burning of residues

From Figure 4 it can be seen that plot BA in the months following burning had a larger surface runoff than plot RA in the control forest, even though plot BA is both less steep and shorter than plot RA. This period was equivalent to the period right after burning when all vegetation was killed and maximum loss of transpiration and interception could be expected. A few months later, as vegetation colonization started to cover the catchment, plot BA had less response in surface runoff than plot RA at major storms (Figure 4).

This observed short lived increase in surface runoff due to the burning was not large and could not account for the double increase in stream runoff in the burned and tractor cleared catchment W5 compared to W4 with no burning and minimum soil disturbance. Rather, like the case for plot MA in W4, the troughflow mechanism in the undisturbed soil could "handle" the increased water volumes without extensive increase in overland flow. Also the streamwater increase during the fist years in W5 occurred as increase in baseflows (Malmer, 1992). However, in the case of the short period with the ash on the soil surface and no vegetative cover and most protecting residues burned, even a small increase in surface wash together with splash erosion could be detrimental in carrying large amounts of ash and leached nutrients to the stream (cf. Malmer and Grip, *in press*).

As pointed out above, extensive surface erosion occurred in plot BA at the first moderate rains after burning, while no erosion could be detected at the same events in the other plots. During 3 days the sediment loss was double the yearly loss in control forest. A very blackish color of collected water and sediments observed also indicated that it was the nutrient rich ash that was lost from the slope. In a couple of weeks a few cm thick layer of ash on the slope surface was washed away along the slope and possibly also down into the pore system of the soil. As indicated by the color of the mineral soil shining through, removal of ash was most effective on convex surfaces. Rill erosion were also observed in the ash, especially on concave surfaces where ash accumulated before reactivation in the transport downslope. Studies of shifting cultivation has indicated that loss of ash by wind may be high (Toky and Ramakrishnan, 1981). In this study wind transport of ash was not observed, but the burned catchment (W5) was rather small and protected from wind by surrounding forest.

Still, the total surface erosion recorded after burning was below the median for 18 undisturbed natural forests reported by Wiersum (1984). However, in a discussion on sustainable site production the quality of the lost sediments and implications for leaching is much more interesting than an exercise in comparing the actual erosion with rates of geomorphological denudation (cf. Malmer and Grip, *in press*).

Implications for a theory of slope solute removal.

The solute concentration of different slope flow routes has been shown to be dependant of residence times (Pilgrim et al., 1979; Burt, 1986). Consequently the concentration of throughflow has been shown to increase with contact time (Pilgrim et al., 1979). Northcliff and Thornes (1978) suggested a bi-phasic nature of soil water drainage based on studies on oxisols at reserva Ducke, Amazonas. Sollins and Radulovich (1988) also stressed the importance of preferential water flow to prevent nutrient loss from the soil matrix. By rapid filling and emptying of macropores at rainfall, the bulk of water is transported to the stream without extensive leaching from the soil matrix, explaining very low solute concentrations in streamflow. Later Northcliff and Thornes (1989) also showed soil solute concentration to be a function of pore size, with highest concentrations in the smallest pores.

So far theories and modelling of solute removal from hill slopes has mostly been dealing with weathering and mineral soil ion exchange as sources for ions lost from soil water solutions. Leaching of dissolved elements directly from the mineralization and decomposing organic material is seldom discussed in the case of the undisturbed forest. For the streams of this study Grip et al. (*in press*) showed elements of geochemical origin to be negatively correlated to streamflow in baseflows from W3 (control catchment on 100 areal percent clay top soil), indicating smaller relative volumes of younger water with time since the last rain. On the other hand Ammonia was significantly positively correlated to streamflow in baseflows from both control catchments, indicating delivery of water of shorter transit time to stream baseflow to decrease in stream baseflow recession. Grip et al. (*in press*) connected this to leaching from organic matter and decomposition together with superficial flows as the only means of losing Nitrogen to the stream in the form of Ammonia.

Transferred to the picture of water pathways to the stream discussed above, means that stream baseflow was a mix of water with ionic content with the signature of organic matter and water with ionic content mainly composed by ions from mineral weathering. At high baseflows water of shorter transit time moving in the top soil trough larger pores, without extensive functional contact with smaller pores dominated by geochemical solute and exchange processes, contributed to a larger part of streamwater. At successively lower baseflows, contribution of water with longer transit times, from deeper soil layers with lower hydraulic conductivity became more important. Grip et al. (*in press*) found no significant differences in concentrations between mean stormflows and mean baseflows, but significant relations between different flow rates of baseflow. That also indicated that the quality of high baseflows were better related to stormflows than to low baseflows. Mean concentrations of Ammonia in baseflows were low (seldom > 0.03 mg l^{-1}) from control catchment W3 (Grip et al., *in press*), indicating a tight balance between release and uptake with only small losses from mineralization and organic material to the stream. The non significant differences in element concentrations between stream mean base- and stormflows showed the system to be close to tight also during rains.

In the case of mineralization of residues after clear-felling, speeded by burning, the situation became different. Malmer and Grip (*in press*) showed significantly increased concentrations of Nitrogen, Phosphorous and Potassium in mean stream stormflows in all treated catchments in this study. As the component of slope surface runoff was shown to be kept small on undisturbed soils also after clear-felling, the origin of this concentrated water was shallow troughflow, and in agreement with above preferably in the larger pores. This water acquired dissolved nutrients to high concentrations before and during infiltration trough the logging residues and the litter layer and had the shortest residence time in large pores with comparably low exchange capacity (cf. Northcliff and Thornes, 1989) and short time for biological uptake processes. During storms, mainly the water of the largest pores were released into the stream, and in between storms the most conductive pores were the first to drain to the stream during baseflow recession. As discussed above major rains probably resulted in surface runoff adjacent to the stream. The water passing logging residues or ash in such flows just before joining streamflow of cause also maintained the high concentrations of N, P and K in stream stormflows.

Consequently, the same process that works as a nutrient conserving mechanism under forest cover, may act as a significant means of supplying the stream with substantial amounts of nutrients after treatments (cf. Malmer and Grip, *in press*). As showed by Malmer and Grip (*in press*) nutrient conserving processes were more effective with more vegetation left living (no burning) and less soil disturbance. Compared with above, that may mainly have been an effect of less water supplied to the stream and lower amounts of availability and slower release of nutrients without burning as well as no living vegetation to use released nutrients.

Gully erosion on tractor tracks

Tractor tracks were the only place where gully erosion could be observed. No measurements of surface runoff were made on the disturbed soils of W5, but the mechanism of throughflow is concluded to be practically shut off as indicated by the dramatic reduction in steady state infiltrability, from 154 mm h^{-1} to practically nil (Malmer and Grip, 1990) and hydraulic conductivity (Table II). Instead most water received on the tractor tracks moved downslope as infiltration excess overland flow (cf. plate 6 in Bruijnzeel, 1990). As pointed out above the increase in streamwater was reflected mainly by increases in baseflow during the first time after treatment. Later, in 89/90, Malmer (1992) showed a change of less baseflow and increases in stormflow which was concluded to depend on development of new water ways along the disturbed surfaces. Similar effects of "delayed" runoff increase and increase of the stormflow component was also showed at similar conversion of dipterocarp forest to cocoa and oil palm plantation (Abdul Rahim, 1988; Bruijnzeel, 1989, after DID, 1986). Douglas et al. (1990) also described how the length of ephermal channels were increased with erosion connected to road construction in Ulu Segama, Sabah.

Data presented here (Table III) also show how the gully erosion still was active in late 1990. These gullies now act as stream channels at storms, funneling the overland flow along the tractor tracks faster than in the earliest stage after treatment. Several years after disturbance the erosion on the disturbed surfaces is much less as most of these surfaces now have become sheltered from direct rainsplash by some pioneer vegetation and vegetation expanding above the soil from the sides. However, slope wash and gulling is still active under what may seem lush and green at a first glance. Douglas et al. (1992) also described a similar situation from Ulu Segama, with active sediment transport routes to the stream channel under protective vegetation. In this study also large amounts of sediments was deponated in the lower parts of the disturbed surfaces without being flushed out into the stream (cf. Douglas et al., 1992). Some of these may be reactivated in rare high magnitude rain events to create severe occasional siltation of the stream (Bruijnzeel, 1990; Spencer et al., 1990).

The heavy erosion along the tractor tracks compared with the short lived and small increase in surface erosion on non disturbed and burned soil surfaces verifies that it was the soil disturbance which was the main trigger for the increases in stream suspended sediments (Malmer, 1990). But, again, the quality of sediments from different sources have to be considered if the evaluation of effects concerns future site productivity. Apart from the considerably lowered production from the tractor extracted and burned catchment (Sim and Nykvist, 1991), the effect on runoff and erosion along the tractor tracks seem to be long lived as to infiltrability, runoff and gully erosion. In many discussions the vegetation is said to cover the tracks quite quickly, indicating fast recovery from great erosion hazard. That is also the case after clearing without soil disturbance (eg. Northcliff et al., 1990). However, in this study, coverage was after 4 years in late 1991 still very incomplete in many parts of the tractor tracks. Also, what looked green and lush was still to a great extent vegetation expanding from the side, and almost nothing was rooted in the very tractor track soil, except for some vines and ferns. In late 1991, 4 years after start of treatment (almost half of the calculated rotation time for the forest plantation) the build up of a litter layer and the development of a change in soil structure due to the presence of organic material and activity had not started on most of the tractor tracks. Only tracks with low disturbance (no excavation and only one pass by the tractor) showed some change in that aspect.

Comparison between clear-felling and selective logging

Hydrological and erosional effects of selective logging with road construction and high-lead extraction of dipterocarp forest in the Ulu Segama, Sabah has been described by Douglas et.al. (1990 and 1992), Spencer et al.(1990) and Sinun et al.(1992). Processes leading to increased erosion and stream sediment concentration in that case were very similar to what has been described above. The major difference between clear-felling and selective logging is that there are more vegetation in more layers left on undisturbed soil surfaces and that the areal percentage of soil disturbance may be less after selective logging, especially by using high-lead extraction instead of tractors. Even so, other writers like Abdulhadi et.al. (1981) have reported as much as 30% of bare soil after selective logging in east Kalimantan, Indonesia.

Obviously there will be big differences in soil disturbance and resulting erosion effects depending on type and volume of extraction and degree of planning of the operation as well as with different topographic, edaphic and hydrological conditions. The case of the tractor extraction in this study was a well planned operation with uphill logging (Megahan and Schweithelm, 1983). Less systematic extraction like "timber cruising" and downhill logging would probably have increased the disturbed soil area and the sediments flushed into the stream considerably.

As indicated by the results of this study, the increase in runoff from selective logging will probably be less than from clear-felling because of the larger amount of surviving vegetation. On the other hand, it is also possible that extensive soil disturbance and tractor tracks may increase the stormflow component and prolong the effect of runoff increase after selective logging in the same way as described above due to clear-felling and tractor extraction.

CONCLUSIONS AND IMPLICATIONS FOR FORESTRY

Control forest slope hydrology was dominated by troughflow in permeable top soils, and stream stormflows were mainly fed by short transit time soil water pushed out by translatory flow. Extensive slope surface runoff was not recorded, but saturated overland flow and return flow on gently sloping ground adjacent to the streams was believed to maintain stream stormflows. This mechanism was maintained on undisturbed soils after clear-felling the forest. Extensive slope overland flow did not occur in any investigated clear-felled areas without soil disturbance. In the undisturbed forest were mineralization is evenly distributed in time and space, these slope hydrological processes were nutrient conserving. After clear-felling on the other hand, the shallow troughflow with short transit times contributed to the large losses of dissolved nutrients from decomposing biomass. Only a short increase in slope surface runoff was recorded after burning logging residues. The possibility of a detrimental effect of this little effect of surface runoff is pointed out as it carried nutrient rich ashes downslope. Otherwise surface erosion on undisturbed soils after clear-felling were rather harmless compared with control forest.

On the other hand the erosion along disturbed surfaces like tractor tracks were considerable. The normal process of water movement in the soil to the stream was changed to extensive overland flow creating long lived erosion and prolonged increases in stream runoff and larger stream stormflows. Natural rehabilitation of disturbed soils can take many years, even if their surfaces will be covered by vegetation and splash erosion ceases. Regarding the effects caused by soil disturbance, a clear-felling operation with mechanical extraction is similar to selective logging and comparisons can be made.

The logging, extraction and silvicultural method should be chosen to minimize the area of soil disturbance, to maintain a protective organic layer and to maintain some vegetation to minimize increase in runoff. Any system minimizing disturbance of top soil and burning is to be preferred. The risk for loosing large amounts of nutrients by erosion as well as by leaching from ashes is also talking against the burning of residues after clear-felling. Cable yarding systems can reduce soil disturbance compared to use of crawler tractors, but are in most cases not suitable for selective logging. In the case of saturated surface soils and surface runoff adjacent to the stream, also riparian reserves may be effective to prevent leaching from residues or ash close to the stream. Strip cutting (eg. Ocaña-Vidal, 1992) is another silvicultural system leaving more living vegetation to reduce on site and off site effects. As a whole there still seem to be a lack of relevant equipment developed for the wet tropical environment. Some equipments used are designed for temperate environments and some rather to move soil than for forestry operations (Couper et al., 1981).

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