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HYDROLOGY OF MOIST TROPICAL FORESTS AND EFFECTS OF CONVERSION: A STATE OF KNOWLEDGE REVIEW

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FOREWORD

In spite of valiant efforts of a small band of scientists who have attempted to dispel the misinformation, misinterpretation, misunderstanding and myth about the role of forests with regard to hydrology and erosion, and what happens when the forest is altered or removed, many of these "four M's" still continue to dominate popular and political thinking. This applies especially to tropical humid forests, which seem to automatically put emotion into command over reason. Count myself in that valiant group that includes John Hewlett, Andrew Pearce, Jack Ives, Bruno Messerli, Brian Carson, J.M. Bosch and the "Queensland Mafia" (Don Gilmour, Dave Cassells and Mike Bonell). All have tried to interpret their own research and that of others into management and policy relevant information that is understandable by the non-scientist. Others whose names are not listed above are known in research hydrology and watershed management circles for their quiet and perhaps less pugilistic efforts, and they deserve much credit.

Dr. L.A. (Sampurno) Bruijnzeel has also been a comrade-in-arms in that somewhat embattled group of myth-busters. I say "embattled" advisedly, because many of us have often been accused of selling out to the enemy, by those who are trying valiantly to slow the rate of tropical forest clearance or alteration, and who wish to claim that tropical forest alteration or loss will: cause major floods; automatically result in dramatic increases in erosion that choke rivers, reservoirs and the marine environment with sediment; impoverish the soil; result in lower water yields especially in the dry season due to the loss of the "sponge effect"; lower water tables thus lowering well levels and making springs unreliable; permit landslides where none ever occurred in the natural forest; result in savanna or even in deserts.

Dr. Bruijnzeel entered the battle out of a solid background of field research in the tropics. My first encounter with his work was that fine volume on hydrological and biogeochemical aspects of man-made forests that was based on his dissertation research in Java and published in 1983. We met shortly after that and became fast friends and respectful colleagues. While pursuing solid research activities, mostly in Indonesia, but extending to Sabah in Malaysia, Palawan in the Philippines, and Fiji, he began to produce state-of-knowledge reviews oriented towards those making decisions about land and water management. The first of these appeared in Wallaceana in 1986 and dealt with the environmental impacts of deforestation and forestation in the humid tropics from a watershed perspective. Recently (1989), for the International Center for Integrated Mountain Development (ICIMOD) in Kathmandu, Bruijnzeel with colleague Bremmer did a fine literature review on "Highland-Lowland Interactions in the Ganges Brahmaputra River Basin".

In what follows, as an activity of UNESCO's International Hydrological Programme IV, the author presents a comprehensive review of what is known about humid tropical forest hydrology, nutrient losses and the effects of these forests on climate. There follows an analysis of the effects of forest alteration and conversion on water and soil. In view of the increased interest in tree planting to rehabilitate degraded lands and to perform the function of sequestering carbon, Dr. Bruijnzeel also covers the soil and water impacts of forestation.

Of special relevance to the International Hydrologic Programme is the concluding material that sets forth the major gaps in our knowledge. The author's evident concern for practical problem-solving research, balanced with more fundamental process research, is an excellent fit with the IHP objectives.

This state-of-knowledge review renders obsolete all previous reviews of tropical forest hydrology including my own modest work. Moreover it will stand for many years as the major work of this kind for it is completely up-to-date and comprehensive. Its publication marks significant progress in achieving the objectives formulated at the 1988 International Colloquium on the Development of Hydrologic and Water Management Strategies in the Humid Tropics. It will help to dispel a few more of the "four M's".

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PREFACE

To many, the tropical forest biome is both one of the most intriguing and endangered ecosystems in the world. As such, tropical rain forests are receiving a fair amount of media attention nowadays, both in the popular and scientific press.

Although the ecology (and to me hydrology is another word for applied ecology) of tropical forests constitutes the subject of a host of recent introductory and specialist books, none of these (in my view at least) have presented adequate and up to date descriptions of the hydrological functioning of these forests nor of the environmental effects of converting them to other types of land use.

The present report is an attempt at reviewing the current state of our knowledge of the hydrology of tropical forest before and after various kinds of disturbance, a subject which, according to Dr. Lawrence S. Hamilton, is loaded with "misunderstanding, myth and misinterpretation".

The foundation for the respective chapters of this report was laid in several review papers written for various meetings in the last four years. With the exception of the paper underlying chapter 5, which was prepared fairly recently, all material was updated and extended and sometimes rewritten entirely.

This review would not have been possible in its present format without the help received from a number of colleagues who readily sent in press material or unpublished reports or data. I would especially like to thank Kees Bons (Free University), Antonio Dano (Ecosystems Research and Development Bureau of the Philippines), Dr. Jean-Marie Fritsch (ORSTOM), Prof. Carl F. Jordan (University of Georgia), Anders Malmer (Swedish University of Agricultural Sciences) and Dr. John Roberts (Institute of Hydrology) in this respect.

I am most grateful to Dr. Jack Gladwell of UNESCO's International Hydrological Programme (IHP) for having given me the opportunity to publish the manuscript as part of the Humid Tropics Programme of IHP-IV. Other financial support for the production of the report was provided by the National Committee for IHP of the Netherlands, the International Association of Hydrological Sciences and by the UNESCO-ITC programme "Geo-

information for environmentally sound management of tropical resources" (Hydrology component) within IHP-IV and is gratefully acknowledged. The assistance rendered in this respect by Ir. Henny Colenbrander in his capacity as chairman of the National Committee and Secretary General of IAHS and by Professor Allard M.J. Meijerink (ITC) proved invaluable.

This monograph will be distributed in a circulation of 2500, 500 of which as part of the above-mentioned UNESCO-ITC programme and extended with an appendix on the use of geo-information systems for sound management of natural resources in the tropics.

Special thanks are due to Ir. Hans Hooghart (TNO) for his help with the final preparations for the printer, to Ir. K. Freerk Wiersum (Agricultural University, Wageningen) for the regular supply of often semi-obscure but always highly relevant literature on tropical forest management and agroforestry, and above all to Irene Sieverding for her support and patience (of which I possess relatively little) during the six weeks that this report took shape.

Amsterdam, 28 August 1990 Sampurno Bruijnzeel

1 INTRODUCTION

It has been estimated that the population of the countries within the humid tropics will more than double between 1980 and the year 2000 and by that time will make up almost 50 per cent of a total world population of about 6.5 billion (Gladwell & Bonell 1990). Because of this escalating increase in population there is a consequent rise in demands on water and land resources for food production, often at the expense of the remaining natural vegetation (Lanly 1982).

Estimates of the areal extent of natural forests in the humid tropics and the rate at which these forests are disappearing vary considerably between investigators (Persson 1974; Sommer 1976; Myers 1980; Seiler & Crutzen 1980; Lanly 1982; Lugo & Brown 1982; Melillo et al. 1984; Fearnside 1987). Such discrepancies are partly caused by differences in definitions as to what is meant by "tropical forest" or "deforestation" and partly by the use of different data sets or the scale and type of imagery used in the inventory (Fearnside 1987). However, whether the current alteration rate of tropical forests lies at 9-15 million ha yr^{-1} (Seiler & Crutzen 1980) or even at 24.5 ha yr^{-1} (Myers 1980), there is a general feeling that the disappearance of tropical rain forest constitutes a major environmental problem to mankind (Prance 1986a; Whitmore 1990).

With their unique diversity of plant and animal life, forests in the humid tropics represent an immense source of food, fibre, timber, medicines and fuel for local farmers, hunters and gatherers and, indirectly, city dwellers elsewhere (Lea 1975; Boom 1985). In addition, tropical forests perform a number of other environmental, cultural and spiritual functions to the peoples of the tropics (Jacobs 1988). They are also of concern to the world community as a whole, however, in that they form a significant element of the global carbon budget (McElroy & Wofsy 1986; Crutzen 1987) as well as a major gene pool whose value has only barely begun to be explored, e.g. in the search for pharmaceutics to cure some of mankind's more serious diseases (Boom 1985; Prance 1986b).

In theory, forest land in the humid tropics can be used in various ways. Hadley & Lanly (1983) have recognised three main alternatives. The first is to maintain the forest with little or no disturbance by man for protection purposes. Nature reserves, steep headwater areas of strategic catchments or geologically unstable areas may be included here. A second option is the sustained management of the natural forest for continuous production of wood and other commodities and services such as soil- and water conservation, wildlife, genetic resources and recreation. A third alternative is to clear the forest and to use the land for farming or grazing, plantation forestry, etc. Clearance for settlements, roads, mines and the like also belong to this category.

In view of the rapidly increasing pressures on tropical forest land, large tracts of land will be converted to other uses in the years to come. Clearly, if today's development activities are not to jeopardise the needs of future generations such development must be sustainable, i.e. minimise adverse impacts on the environment.

It is part of the responsibility of the scientific community to help resource managers take the right decisions by providing them with the maximum amount of relevant information "in well-digested and constructed formats" (Gladwell & Bonell 1990). As a first step towards this goal the present report aims to critically review the role of tropical forest with respect to climate, soil and water.

First, a quantitative description is given of the various components of the hydrological cycle in undisturbed moist tropical forest, defined here (Vitousek & Sanford 1986) as (1) lying between 23°N and 23°S, (2) receiving at least 1600 mm of rainfall annually and (3) experiencing no more than four dry (sensu Schmidt & Ferguson (1951), i.e. with rainfall less than 60 mm) months, or (4) experiencing a longer dry season in combination with annual rainfall exceeding 3500 mm and adequate soil moisture storage opportunities (Chapter 2).

This is followed by a discussion of nutrient input-output budgets for a number of tropical forest ecosystems which also examines the commonly held notion of tropical rain forests as having a relatively rich nutrient economy perched on a nutrient poor substrate and only able to maintain themselves via a "tight" nutrient cycle (Chapter 3).

Having presented the forest hydrological "baseline" information in the preceding chapters, the effects of forest disturbance and conversion to other uses are addressed in Chapters 4 (hydrological impacts) and 5 (hydrochemical and soil chemical aspects).

One often reads generalisations that "deforestation" in the tropics results in widespread soil erosion and associated reservoir siltation, in floods, "droughts" and desertification (Daniel & Kulasingam 1974; Goodland & Irwin 1975; Eckholm 1976; Bowonder 1982; Sharp & Sharp 1982; Myers 1986, etc.). However, as pointed out by Hamilton (1987), the term "deforestation " is used so ambiguously that it has become rather meaningless as a descriptor of land-use change. As such, before environmental impacts of tropical "deforestation" can be evaluated properly, one needs to precisely describe the land-use change c.q. activity under consideration (see Hamilton & King (1983) for numerous examples).

It is helpful in this respect to distinguish three levels of intensity of disturbance, viz. low, intermediate and high (Jordan 1985). The first category includes such small-scale and short-lived events as natural tree falls and small clearings. Selective logging, forest fire and shifting cultivation may rank under the second heading whereas forest clearing and conversion to pasture, extractive tree crops, forest plantations or permanent annual cropping classify as high-intensity types of disturbance.

Naturally, given the extent of severely eroding, hydrologically and ecologically disrupted lands in the humid tropics, calls for massive reforestation programmes are becoming more frequent (Eckholm 1976; Postel & Heise 1988). Yet it may be premature to expect that reforesting degraded lands will fully restore the original hydrological regime (Hamilton & King 1983) whilst in addition there may be productivity problems if these forest plantations are to be harvested over several rotations (Chijioke 1980). Chapters 4 and 5 therefore also review the available information with respect to the hydrological and soil fertility aspects of "forestation"; cf. Wiersum 1984b).

Finally, Chapter 6 summarises the current scientific consensus about the hydrological role of tropical forest, indicates gaps in our knowledge and offers suggestions for further research. The report concludes with a list of about 700 literature references.

2 THE HYDROLOGICAL CYCLE IN MOIST TROPICAL FOREST

2.1 General description of processes and variables

Although most readers will be familiar with the general outline of the forest hydrological cycle, there still exists a fair deal of semantic confusion about such terms as "runoff", "interception loss", etc. Therefore, a brief description of the forest hydrological system is in order (Figure 1). Since forest ecosystems are generally part of larger landscape units drained by one or more streams, this is followed by a discussion of how different parts of a drainage basin are linked hydrologically (Figure 2).

Apart from specific locations, such as coastal fog belts or cloud belts, rain is the precipitation input to forests in the humid tropics. A small part reaches the forest floor directly without touching the canopy: the so-called "direct" or "free" throughfall (Rutter et al. 1971). Of the rain that strikes the vegetation a substantial portion is intercepted by the canopy and evaporates back into the atmosphere during and immediately after the storm (Bruijnzeel & Wiersum 1987).

Many hydrologists like to refer to this component as a "loss". However, if it is assumed that leaves covered with a film of water do not transpire (the energy necessary for transpiration largely being consumed to evaporate the water on the leaves), then while intercepted water is being evaporated there will be a saving in water taken up for transpiration (Rutter 1975). Burgy & Pomeroy (1958) therefore introduced the distinction between "gross" (i.e. incident rainfall minus net rainfall) and "net" interception loss (i.e. the gross loss minus the saving in transpired water). Since the latter is difficult to determine and virtually no estimates have been published for tropical conditions, the distinction does little more than creating confusion. In the present review rainfall interception is taken as gross rainfall minus net rainfall (see below).

The remainder of the rain reaches the forest floor as crown drip and via branches and trunks as stemflow after the respective storage capacities of the canopy and the trunks have been filled (Leyton et al. 1967). The sum of direct throughfall, crown drip and stemflow is commonly called net precipitation (Helvey & Patric 1965).

Although direct throughfall and crown drip cannot be determined separately in the field, the distinction between the two is useful in modelling the interception process (Rutter et al. 1971; Gash 1979).

Normally, amounts of precipitation measured within tropical forests are substantially less than outside, but in coastal fog belts and montane cloud belts the reverse sometimes applies because of "occult precipitation", the stripping of fog or clouds by the vegetation (Zadroga 1981; Stadtmuller 1987).

The rainfall arriving at the soil surface encounters a filter that determines the path to reach the stream channel (Figure 2). The water in the various pathways may be characterised by solute concentrations which reflect differences in residence time (Bruijnzeel 1983b) and as such the prevailing hydrological pattern influences amounts of nutrients leached from the system (Burt 1986). This in turn may influence forest structure (e.g. Lescure & Boulet 1985) and determine the fate of the land following forest clearing (Nortcliff & Thornes 1981).

Since precipitation inputs are discrete and sometimes, especially in the more seasonal tropics, widely separated in time, streamflows exhibit periods of suddenly increased discharge associated with rainfall and longer periods of slowly decreasing flow of water stored in the catchment. The immediacy of streamflow response suggests that part of the rainfall will follow a rapid route to the stream channel, thereby producing so-called "quickflow". The water travelling more slowly emerges as "baseflow" (Ward 1984).

If rainfall intensities below the forest exceed the infiltration capacity of the soil, the unabsorbed excess runs off as "Hortonian" or "infiltration excess" overland flow (Horton 1933; flow path Q_o in Figure 2a). The remainder infiltrates into the soil and, depending on vertical and lateral hydraulic conductivities, local soil moisture patterns and slope steepness, may take one of several routes to the stream channel (Figure 2a).

In the (relatively rare) case of deep, permeable and uniform deposits the water will tend to travel mainly vertically to the zone of saturation and hence follow a curving path to the stream channel (flow path Q_g in Figure 2a). Generally, however, permeability decreases with depth. Most of the water then percolates vertically until it meets an obstruction

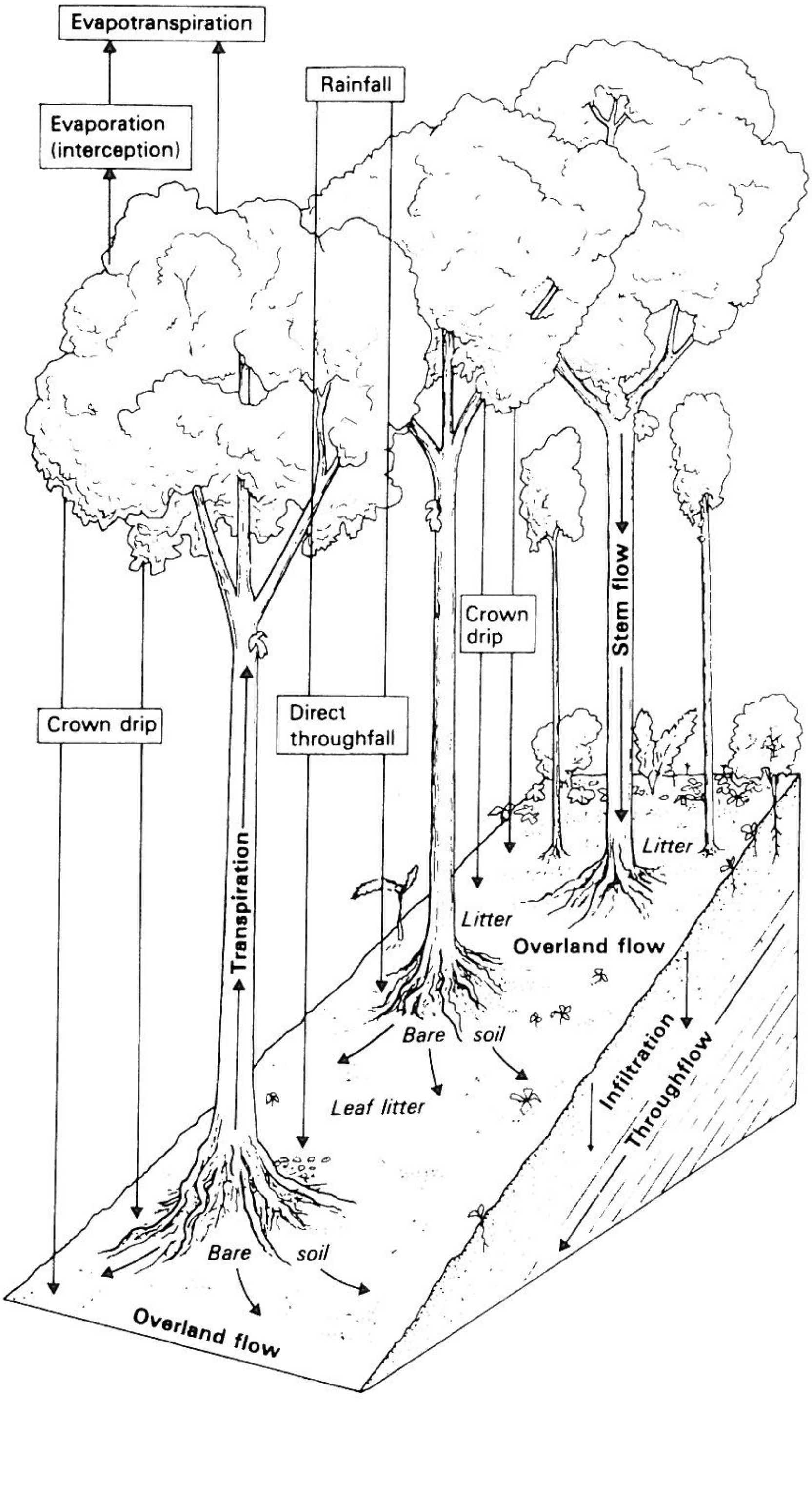

Figure 1. The hillslope forest hydrological cycle (adapted from Douglas 1977).

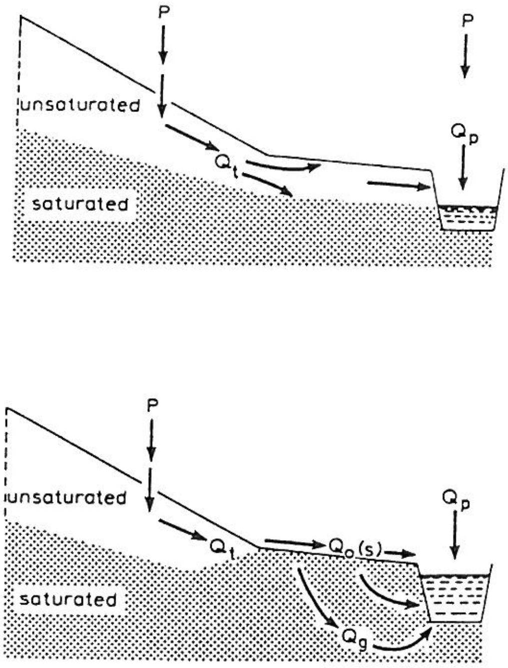

Figure 2. Flow paths of the sources of streamflow (Ward 1984). Q_p is direct precipitation on the water surface, Q_o is "Hortonian" overland flow, Q_t is throughflow, Q_g is groundwater flow and $Q_o(s)$ is saturation overland flow.

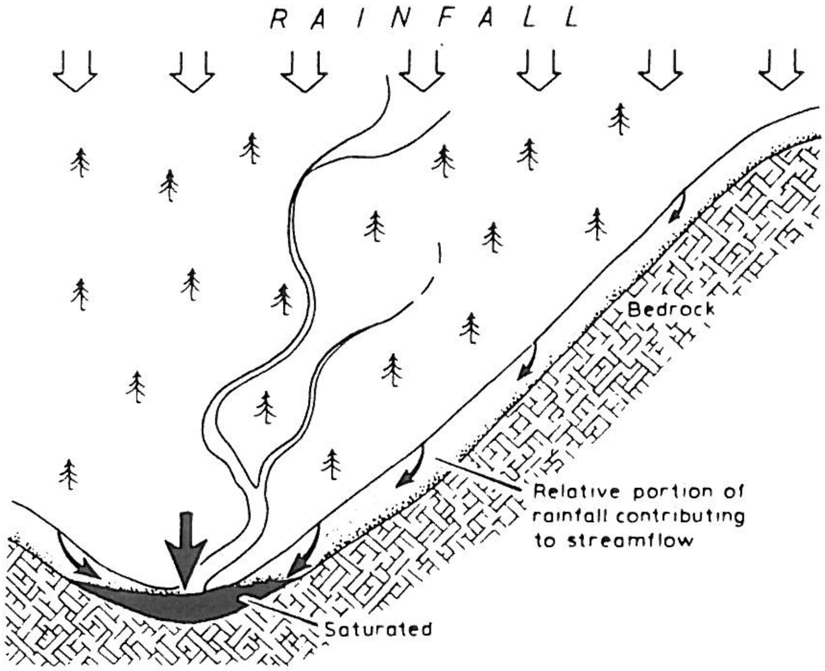

Figure 3. The relative contributions of rainfall to streamflow (after Ward (1984), based on an original diagram in Hewlett (1961)).

such as a clayey B-horizon or bedrock, and is deflected laterally (e.g. Weyman 1973; Guehl 1983). Usually this lateral flow in the soil profile is referred to as "throughflow" (Kirkby & Chorley 1967; flow path Q_t in Figure 2a, see also Figure 1).

Throughflow generally travels relatively slowly through the soil matrix, feeding near-saturated sections around stream channels and in topographic depressions, thereby maintaining the baseflow of the stream (Hewlett & Hibbert 1963; Figure 3). These near-saturated zones in a catchment often act as the major source of quickflow during rainstorms (Dietrich et al. 1982; Nortcliff & Thornes 1984).

The mechanism of quickflow production reflects the prevailing climatic, geomorphological and pedological setting (Walsh 1980; Ward 1984; Burt & Butcher 1985). In case of the situation depicted in Figure 2b (wide valley bottom) quickflow is generated through the formation of what has been called a "riparian groundwater ridge" (Ward 1984), which may rise to the surface and induce so-called "saturation overland flow" (Dunne 1978; flow path $Q_o(s)$ in Figure 2b). In such cases a significant portion of the quickflow will consist of freshly fallen water with relatively low solute concentrations (Bruijnzeel 1983b) and stormflow patterns will reflect differences in rainfall intensity as they occur (Dunne 1978; Elsenbeer & Cassel 1990; Figure 4).

However, where deep permeable soils overlie impermeable bedrock and where steep hillslopes border a narrow flood plain, there will be little scope for saturation overland flow, neither in the valley bottoms nor on the hillsides themselves. Rather, quickflow will then be dominated by (rapid) throughflow contributions (Harr 1977; Dunne 1978; Cales 1982). Depending on the depth and initial moisture status of the soil and the size of the storm a peak in streamflow may occur shortly after the storm or up to several days later (Hewlett & Nutter 1970; Sklash et al. 1986; Figure 4).

To explain the immediacy of streamflow response to rainfall in areas without appreciable overland flow of any type it has been suggested that part of the throughflow travels through the upper soil horizons quickly enough to reach the stream channel during the storm (hence the term "subsurface stormflow"). Decayed root channels, animal burrows and other "macropores" have been advocated as conducts for this type of flow (e.g.

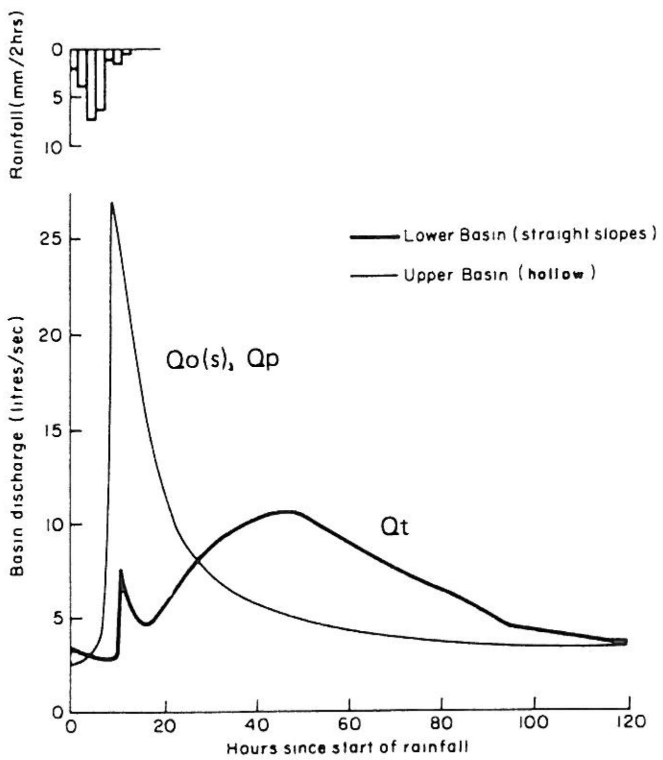

Figure 4. Stormflow hydrographs from two areas with contrasting topography within the East-Twin basin (0.2 km^2), England (after Calver et al. (1972)).

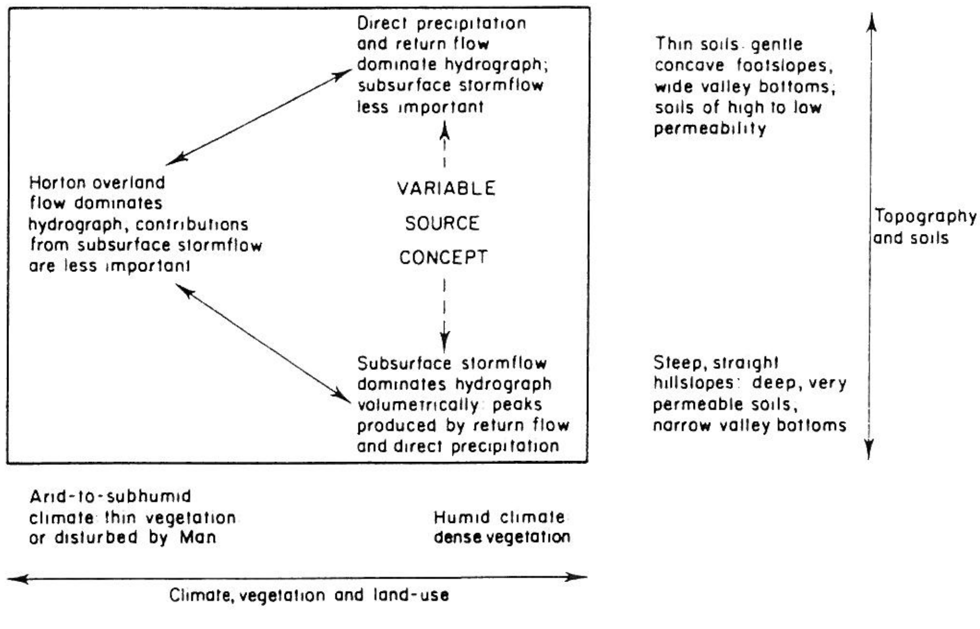

Figure 5. Schematic representation of the occurrence of various streamflow generating processes in relation to their major controls (after Dunne (1978); "direct precipitation and return flow" equivalent to saturation overland flow).

Whipkey 1965; Mosley 1982a). Although this may be true in certain cases, e.g. where subsurface pipes have developed (Jones 1981; Elsenbeer & Cassel 1990), rates of water movement through soils are generally far too slow to enable "new" rainfall to reach the stream during a storm event as throughflow (Dunne 1978). Therefore, Hewlett & Hibbert (1967) advanced the concept of "translatory flow", a "push-through" mechanism whereby each new addition of rain to a hillside displaces an approximately equivalent amount of "old" water, thus causing the oldest water to exit from the bottom of the slope into the stream (Figure 3).

Despite the importance of macropores in soil hydrology (White 1985; Sollins 1989; Nortcliff & Thornes 1989), recent work on stormflow generation using natural isotope and chemical tracers in areas of high rainfall, short and steep slopes and highly permeable soils (i.e. the ideal conditions for the generation of rapid subsurface flow), has shown the importance of the push-through mechanism under such conditions (Pearce et al. 1986; Sklash et al. 1986).

Naturally, an equivalent displacement of stored soil water by "new" rainfall can only be expected if the moisture storage capacity of the soil mantle is nearly full (i.e. wetter than field capacity). In drier conditions rainfall additions will be used to "top-up" the soil moisture store rather than to displace existing moisture. This of course implies that the mechanism will be effective most frequently after a period of rain and/or on the lower and moister parts of the slopes (Ward 1984).

As shown in Figure 4, peaks produced by some form of overland flow tend to be much more pronounced than those generated by subsurface flow. It is especially this shift from subsurface flow- to Horton overland flow dominated stormflow that often accompanies certain changes in land use which produces some of the problems to be discussed in Chapters 4 and 5.

Figure 5 summarises the occurrence of the various streamflow generating processes in relation to their major controls. As emphasised by Dunne (1978), the various modes of stormflow should be seen as complementary rather than contradictory.

Naturally, not all of the water infiltrating into the soil emerges as streamflow, a large part of it is taken up by the forest and returns to the atmosphere via transpiration. In the present context the term forest evapotranspiration (ET) will be used to denote the sum of transpiration (i.e. evaporation from a dry canopy) and interception (i.e. evaporation from a wet canopy). Evaporation from the litter and soil surface has been shown to be minor in humid tropical forests (Jordan & Heuveldop 1980; Roche 1982a,b).

2.2 Hillslope hydrological patterns

Generally speaking, infiltration capacities of forest soils are high (Douglas 1977; Pritchett 1979) and infiltration excess overland flow in forests is therefore a rare phenomenon, even under intense rainfall. However, where the soil becomes exposed through treefall or landslips on steep slopes effects of overland flow may be important (Ruxton 1967; Imeson & Vis 1982). The latter authors found the amount of bare soil to vary greatly within and between forest types in the Colombian Andes. They reported an average value of 6 per cent of bare soil (range 0-21) with a slight trend for lower values to occur occur in montane forests.

A rather extreme case has been reported by Herwitz (1986a) for a forest in Queensland with an annual rainfall of about 6500 mm. There the combination of high intensity rainfall and its funnelling by the trees produced large volumes of water around the trunk bases which often exceeded local infiltration capacities with the result that water flowed downhill over the surface for varying distances. However, work in other tropical forests has shown that this type of runoff is usually less than 1 per cent of the rainfall (Nortcliff et al. 1979; Lundgren 1980; Roose 1981; Cales 1982; Hatch 1983; Coelho Netto 1987; Vis 1989).

On the other hand, on forested slopes in the tropics that were underlain by soils with an impermeable horizon relatively close to the surface, substantial (up to 47 per cent of total streamflow) volumes of saturation overland flow have been reported (Bonell & Gilmour 1978; Walsh 1980; Dietrich et al. 1982; Sarrailh 1983; Elsenbeer & Cassel 1990). Therefore, for a given rainfall regime hillslope flow patterns are very much dependent on the nature of the substrate (Figure 5).

At one end of the spectrum there are those substrates which are sufficiently permeable to prevent any type of overland flow from occurring more than occasionally on the hillside (e.g. Nortcliff et al. 1979; Walsh 1980; Roose 1981; Bruijnzeel 1983b; Coelho Netto 1987). In the case of narrow valley bottoms (usually in steep terrain) streamflow during storms will then be dominated by rapid throughflow/translatory flow contributions from upslope (Walsh 1980; Bruijnzeel 1983b; Vis 1989) or, in the case of wider valley bottoms, by locally generated saturation overland flow (e.g. Nortcliff & Thornes 1984; Poels 1987; see also Figure 2b).

At the other extreme, for soils which have a sudden decrease in permeability at shallow depths, there is the widespread occurrence of saturation overland flow on the slopes (as opposed to the valley bottom) dominating the stream's storm hydrograph, often in conjunction with pipe flow (Bonell & Gilmour 1978; Bonell et al. 1981; Dietrich et al. 1982; Elsenbeer & Cassel 1990).

Depending on the geological make-up of an area a whole array of hillslope hydrological situations (freely draining soils vs. soils of restricted drainage) may be encountered (see for example the work of Cales (1982), Guehl (1983), Lescure & Boulet (1985) and Fritsch et al. (1987) in French Guyana and that of Ternan et al. (1987) in Grenada). Naturally, such variations should be taken into account if problems upon "developing" these areas are to be avoided (Cassells et al. 1984; Ternan et al. 1989).

2.3 Components of the hydrological cycle in moist tropical forest

2.3.1 Precipitation

For a long time there have been strongly held opinions that the upstroke of moisture into the atmosphere associated with evaporation provides water for re-precipitation on the surface from which the evaporation took place (see Penman (1963) for early references on the subject). Stated more explicitly, is the high rainfall experienced by rain (sic!) forested areas such as the Congo and Amazon basins a result of the presence of the forest, or is the forest a consequence of the high rainfall?

Penman (1963) re-examined data on rainfall, streamflow and evaporation presented by Bernard (1945) for different vegetation zones within the Congo basin and observed that the mean annual evaporation was very nearly constant and independent of the type of vegetation in spite of a wide variation in rainfall, suggesting that the amount of rain determined the type of cover rather than the converse. Penman's conclusion was confirmed by reviews of the subject by Golding (1970) and Pereira (1973) who stated that there is as yet no conclusive evidence for forests in the humid zones of the world to have a significant effect on the amounts of rain falling on extensive regions or local areas. Indeed, most of the "evidence" for increased rainfall in forested areas could be dismissed as an artefact of the data related to differences in rainfall catch associated with wind effects around the rain gauges (Lee 1978).

More recently, Salati et al. (1979) postulated that the very small inland gradient of the oxygen-18 content of precipitation in the Amazon basin was the result of a large contribution of re-evaporated moisture to the basin's water balance. A descriptive model was developed which subdivided the Central Amazon basin into 3° longitudinal segments. The observed inland gradient in isotopic concentration of rainfall could then reasonably be explained by assuming that in each segment rainfall was derived for one half from evapotranspirational recycling within the segment itself, with the remainder produced from water vapour derived from the neighbouring segment to the east (Dall'Olio et al. 1979).

The geoclimatic setting of the Amazon basin, a horse-shoe shaped plain open to the moisture-bearing eastern trade winds and effectively sheltered by mountains and high plateaus to the west, north and south, is indeed conducive to this hypothesis (cf. Plate 1). Additional support comes from measurements of regional vapour fluxes which suggest that the input of oceanic moisture account for about half of the region's total precipitation (Marques et al. 1980a,b). Also, the best estimates of forest evapotranspiration available for the region confirm that on an annual basis about 50 per cent of incident rainfall is re-evaporated into the atmosphere (Shuttleworth 1988a).

However, one may wonder whether the Amazonian setting is all that different from that encountered in the Congo basin where according to Bernard (1945) evaporation was independent of rainfall. Interestingly Bernard himself hypothesised about two component cycles within the overall hydrological cycle for the basin: an "interior" cycle in which all evaporated moisture would become precipitation and an "exterior" one in which the remainder of the rainfall was supplied by moisture from the Atlantic, although he had no way of proving this. The similarity with Salati's hypothesis for the Amazon is striking indeed and a regional survey of isotope contents of rainfall and streamflow in the Congo could be a fruitful exercise.

Although the recycling hypothesis has been presented as an actual description of the hydrology of the Amazon basin in about every recent account on the subject (e.g. Salati et al. 1983; Salati & Vose 1984; Salati & Marquez 1984; Salati et al. 1986; Salati 1987; Leopoldo et al. 1987), others (e.g. Pearce & Hamilton 1987; Rodda 1987) have pointed out that the basic hydrological and meteorological data for the Amazon region are still so inadequate that any conclusions must be viewed with caution.

Recent support for the recycling hypothesis, however, has come from computer simulation studies "predicting" the climatic effects of a large-scale conversion of Amazonian forest to degraded grassland (Dickinson & Henderson-Sellers 1988; Shukla et al. 1990; see section 4.2 for details).

Undisputed, however, is the fact that the vegetation occurring in areas subjected to frequent or persistent fog or clouds, such as coastal fog belts and at high elevations, is capable of capturing atmospheric moisture through the process of "cloud stripping". These "cloud forests" (Stadtmuller 1987; Plate 2) cover about 500,000 km^2 (Persson 1974).

Given the right conditions, increases in total precipitation through cloud stripping can be sizeable, especially so for isolated single trees or rows of trees (Ekern 1964; Juvik & Ekern 1978; Kashiyama 1956). The effect is somewhat less in the case of closed forest due to the mutual sheltering of trees but hundreds of mm yr-1 may be contributed by occult precipitation in forests subjected to persistent wind-driven fog and/or clouds (Stadtmuller 1987). Typical values for the wet tropics range between 4 and 18 per cent of ordinary rainfall (Hermann 1970; Weaver 1972; Caceres 1981; Vis 1986; Cavelier & Goldstein 1989) to over 100 per cent under more seasonal conditions (Vogelmann 1973; Cavelier & Goldstein 1989).

2.3.2 Throughfall and stemflow

No other component of the hydrological cycle in tropical forests has received so much attention as the measurement of net precipitation. Yet, results from over 100 "tropical" interception studies, more than seventy conducted in natural forests, are quite diverse and differ greatly in their reliability for a number of reasons, mainly climatic, vegetative and procedural (L.A. Bruijnzeel, unpublished manuscript; cf. a similar conclusion about investigations of fine litterfall in tropical forests by Proctor 1983).

The highly complex nature of the processes involved (Leonard 1967; Jackson 1975) and the high spatial and sometimes temporal heterogeneity of tropical forest canopies (Figure 7) require elaborate sampling designs which up to now have been rarely used (Lloyd & Marques-Filho 1988). Out of the 77 throughfall (Tf) studies listed by Bruijnzeel (unpublished manuscript) only 20 were considered to have sampled net precipitation more or less adequately (i.e. with confidence limits of 10 to 15 per cent). Whereas statistical evaluations of the number of gauges required for a certain level of precision have been available for deciduous (Czarnowski & Olszewski 1970) and coniferous (Kimmins 1973) forests in the temperate zone, such information was lacking for tropical rain forests until recently (Lloyd and Marques-Filho 1988).

Figure 6 suggests that to estimate mean Tf in a tropical lowland rain forest with a confidence limit of 10 per cent, one would need about forty (standard type) rain gauges in a fixed arrangement. Thus far only Pereira (1952), working in a montane (bamboo) forest in Kenya, has used such a large number of gauges. However, the reliability of the estimate improves rapidly if the gauges are relocated regularly to random locations on the forest floor (Lloyd & Marques-Filho 1988; Figure 6).

Few investigators of Tf in tropical forests have practised gauge relocation (e.g. Ducrey & Finkelstein 1983; Collinet et al. 1984; Vis 1986; Lloyd & Marques-Filho 1988; Hutjes et al. 1990) and it is difficult to assess the deviation from the "true" mean for the numerous fixed-gauge studies. Some workers have approached the problem of spatial heterogeneity by increasing their sampling surface through the use of large metal plates (Gonggrijp 1941b), elongated gutters (Clements & Colon 1975; Jordan & Heuveldop 1981) or plastic sheets (Calder et al. 1986b). However, the use of a large collecting surface per se is no guarantee of a good estimate. This was demonstrated indirectly by Bruijnzeel & Wiersum (1987), who used ten to twelve trough-type gauges (with an equivalent surface of 140-168 standard gauges) to estimate Tf in a young plantation of *Acacia auriculiformis* A. Cunn. in Indonesia. Their measurements were made during two consecutive rainy seasons with fixed trough arrangements that differed between both periods as the troughs were removed during the dry season. Although both estimates of seasonal Tf had standard errors of

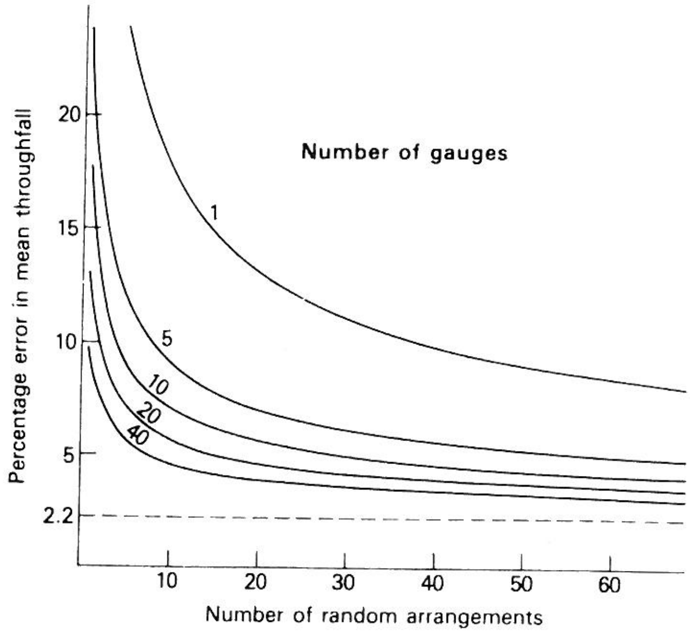

Figure 6. Probable standard errors of the mean of measured throughfall in lowland rain forest near Manaus, Brazil, for increasing sample size and numbers of gauge relocations (Lloyd & Marques 1988).

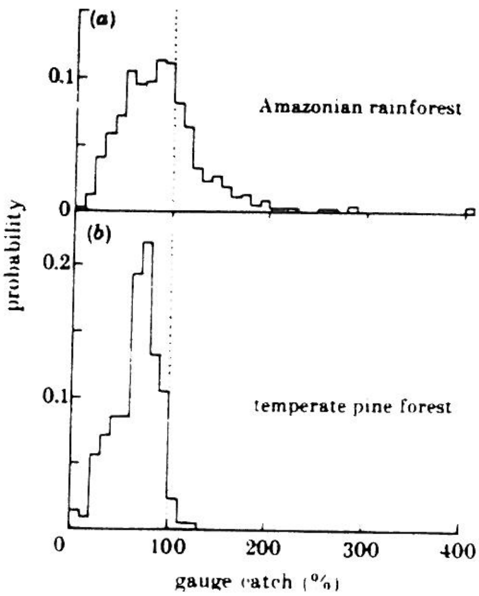

Figure 7. Probability distribution of throughfall gauge catch in a random grid expressed as a percentage of coincident gross rainfall for (a) Amazonian rain forest and (b) pine plantation in the U.K. (after Lloyd & Marques-Filho 1988).

less than 5 per cent, the means were significantly ($p < 0.001$) different (at 81 vs. 75 per cent of incident rainfall). Since the difference was only partially explicable in terms of increases in canopy storage capacity or storm patterns, it had to be concluded that the two trough configurations sampled the spatial variation in Tf differently.

Therefore, the majority of Tf studies in tropical forests must be considered as more or less inadequate. Interestingly, often the highest Tf percentages have been recorded by those studies in which the gauges were randomly relocated at regular time intervals (Bruijnzeel, unpublished manuscript), probably because of the inclusion of a more representative number of "drip points" where Tf is concentrated and exceeds incident precipitation (cf. Figure 7; Lloyd & Marques-Filho 1988).

Choosing only those investigations which both lasted at least a year (to account for seasonal variations in storm patterns and/or vegetation status) and which used a gauge-relocation technique or a fixed arrangement of at least twenty gauges, yielded an average annual Tf of 85 (range 77 - 93) per cent of incident rainfall for lowland forests ($n = 13$) and of 81 (range 75 - 86) per cent for montane forests ($n = 6$) (Bruijnzeel, unpublished manuscript). In light of the above remarks these average estimates should be regarded as conservative.

Incidently, one of the more important ecological implications of the apparent underestimation of Tf in tropical forests by most investigations is that the net retention of nutrients by the canopy upon the passage of rain as reported by some (e.g. Manokaran 1980) may be an artefact of this underestimation. Although net precipitation estimates for the San Carlos forest (where the idea of "nutrient scavenging" by the forest canopy as an adaptation to oligotrophic conditions originated (Jordan et al. 1980)) seem reliable, the quality of the analytical data is questionable (Galloway et al. 1982; Vitousek & Sanford 1986). Indeed, the majority of studies of throughfall chemistry in tropical forests have reported net leaching of nutrients from the canopy regardless of soil fertility (Vitousek & Sanford 1986). There is a need for more good studies of throughfall quality in the tropics.

For stemflow in lowland tropical forests there are many reports that it accounts for about 1 to 2 per cent of incident rainfall (Bruijnzeel, unpublished manuscript). Overall contributions of stemflow to soil water may be small, but it is important compared with the amount of rainfall intercepted by the forest. In addition, stemflow carries amounts of nutrients to the bases of individual trees that are too large to be ignored (Herwitz 1986b).

Again, stemflow measurements should take into account the large spatial variability of tropical forests (Lloyd & Marques-Filho 1988), especially when attempting to determine amounts of nutrients carried to the forest floor via stemflow (Herwitz 1986b). Stemflow is usually measured volumetrically and the amounts are subsequently converted to millimetres of water by dividing them by projected crown area (Freise 1936). However, given the difficulties associated with estimating crown areas in tropical rain forests it may be more advisable to sample every tree within a sufficiently large plot and to divide the total stemflow volume by the area of the plot to obtain an areally weighted average (Helvey & Patric 1965; Frangi & Lugo 1985).

Several investigators reported stemflow from large-diameter trees to be less than for smaller-stemmed trees (Weaver 1972; Jordan 1978; Lloyd & Marques Filho 1988). This may be ascribed to differences in branching patterns as well as to the fact that drip from the higher trees may become funnelled again upon hitting trees lower down. This could possibly enhance nutrient availability for smaller trees and could perhaps be part of an explanation for the many-stemmed pole habit of trees in many heath forests or montane forests where nutrients, especially nitrogen (Vitousek & Sanford 1986; Marrs et al. 1988) might be limiting.

A related aspect is the erosive power of stemflow and crowndrip. Since stemflow represents a concentration of water at the base of the stem, in certain climatic and topographic conditions it may take on dramatic proportions (Herwitz 1986a; cf. section 2.2) capable of washing away litter and topsoil (Ruxton 1967; Douglas 1977).

Several recent studies suggest that - contrary to common belief (e.g. UNESCO 1978) - the kinetic energy of throughfall in natural and man-made forests in the tropics is higher because of increases in drop size than for rainfall in the open (Wiersum 1983; Vis 1986; Brandt 1988). This change in drop size was shown to be greater under a multiple canopy than under a single canopy, but the lack of falling height to attain sufficient velocity in the former case meant that kinetic energy gained through the drop-size changes was lost in low terminal velocities (Brandt 1988).

The amount of splash erosion induced by throughfall hitting the forest floor is governed by the degree to which the ground surface is covered by litter or undergrowth rather than by the kinetic energy (erosive power) of the rain drops (Wiersum 1983). Williamson (1981) suggested that the "drip tips" which are so often observed on the leaves of tropical plants tend to produce smaller drops, thereby decreasing the risk of splash erosion. Later, Williamson et al. (1983) tentatively explained the vertical distribution of drip tips in the canopy (Richards 1952) in these terms. Phytomorphological and erosion surveys at a number of contrasting sites, together with further experimental work along lines indicated by Wiersum (1983) and Brandt (1988) could throw more light on this problem.

2.3.3. Evapotranspiration

Shuttleworth (1979) and Stewart (1984) have reviewed the techniques for estimating evapotranspiration (ET). A general distinction can be made between water balance methods and micrometeorological techniques. The former find ET by difference and involve measurements of precipitation, streamflow or drainage and changes in soil moisture and groundwater storages and are commonly applied to (presumably watertight) catchment areas (Lee 1970). Most micrometeorological methods on the other hand, require sophisticated instrumentation at various levels within and above the forest canopy and have therefore been much less widely used (Shuttleworth et al. 1984). The majority of published estimates for ET in moist tropical forests has been made by the water balance method on catchment areas ranging in size from a few hectares to the entire Congo (Bernard 1945) and Amazon (Leopoldo et al. 1987) basins.

Catchment water balance studies, however, suffer from the uncertainty that ungauged, subterranean flow (either into or out of the catchment) may constitute a significant part of the total movement of water. In many small headwater catchment areas streams have not yet cut through the entire weathering mantle which may reach considerable depths in the tropics (Burnham 1989; cf. Chapter 3) whereas larger streams may lose substantial amounts of water to their floodplains. Naturally, bedrock underlying valley fills or weathering mantles may be leaky itself, volcanic (Gonggrijp

1941b; Rijsdijk & Bruijnzeel 1990) and karstic (Walsh 1982; Crowther 1987b) terrains being especially notorious in this respect. In case of ungauged leakage ET will be overestimated proportionately.

Also, catchment studies are unable to provide short-term estimates of evaporation (i.e. for periods less than a week) or information on the relative importance of the various components of evaporation ("black box" approach). Finally, areal precipitation estimates are often unreliable for larger tropical basins, especially forested ones (Aitken et al. 1972; cf. Rodda 1987). Within the limitations of the method, theoretically the best estimates could be obtained from carefully selected catchments of relatively small size (up to a few km^2) monitored for a number of years to account for climatic variability. However, even under these conditions a precision of 15 per cent for ET is already difficult to achieve (Lee 1970; Bruijnzeel 1983a).

In light of the shortcomings of the water balance technique, several recent studies of tropical forest evaporation have attempted an alternative, converse, approach in which evaporation from wet and dry canopies are treated separately and the rate of water vapour transfer into the atmosphere is integrated subsequently (Calder et al. 1986b; Shuttleworth 1988a). Shuttleworth et al. (1984) reported systematic errors of 5 to 10 per cent for energy flux measurements above a forest in central Amazonia whilst Calder et al. (1986b) estimated a precision of about 16 per cent for the value of ET derived for an old secondary forest in West Java.

Sometimes one may want information on ET for a specific part of a forest (e.g. on exposed ridges or on a floodplain) rather than the lumped estimates given by the above approaches. This then introduces the practical problem of estimating the soil drainage component, which cannot be evaluated by regular measurements of changes in soil moisture storage alone (Huttel 1975). This component may be measured by means of a (large) lysimeter or estimated by the hydraulic conductivity - potential gradient method, which in wet climates involves continuous or at least daily observations of soil water tensions and knowledge of the associated changes in unsaturated hydraulic conductivity (Cooper 1979; Bruijnzeel 1987). Both techniques have serious limitations (Shuttleworth 1979).

Results for lowland forests

Published results are quite variable. For example, estimated annual values of ET of lowland and hill dipterocarp forests on granitic substrates in Peninsular Malaysia - all receiving more than 2000 mm of rain annually - range from about 1000 mm (Low & Goh 1972) to almost 1800 mm (Abdul Rahim & Baharuddin 1986). Similar variations could be quoted for West Africa or Amazonia (see footnotes in Table 3, Chapter 3). Much of this variation is explicable in terms of basin leakage or methodology.

Values for a number of studies are given in Figure 8. Additional information is presented in Table 1. The scatter of the points in Figure 8 is partly due to the inclusion of a few studies, such as those referred to as * 9, ▲ 3 and ▲ 4, which were carried out during dry years. The wettest location of all (the Guma basin in Sierra Leone, ▲ 6) experiences a severe dry season of five months, despite a high annual precipitation, which may account for its relatively low ET value (Ledger 1975). A somewhat similar situation has been reported for a small catchment covered with (mature?) secondary forest in the Philippines (□ 4; Baconguis 1980).

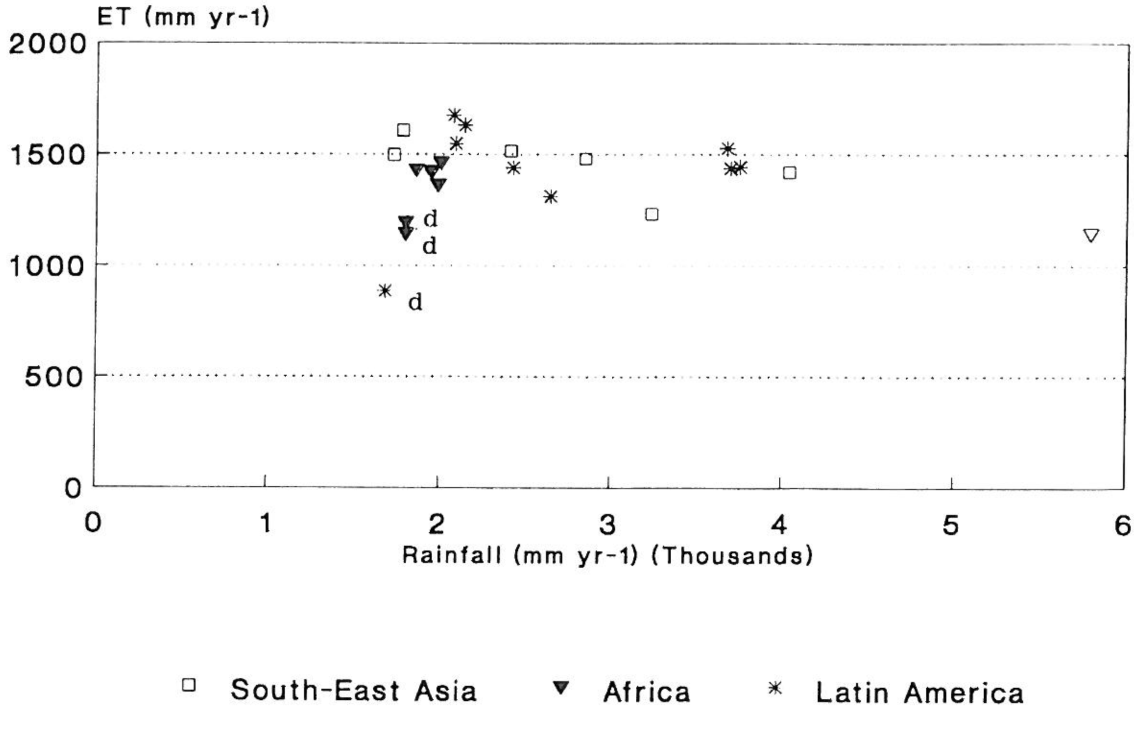

Figure 8. Annual ET versus precipitation in tropical lowland forests. d is dry year (cf. Table 1).

TABLE 1. Annual evapotranspiration (ET) versus precipitation (P) for selected tropical lowland forests

Location	P (mm yr^{-1})	ET (mm yr^{-1})	Length of observation (yr)
Latin America (*)			
(1) Ducke Reserve, Brazil[1]*	2648	1311	2
(2) idem[2]^	2075	1675	1
(3) Bacio Modelo, Brazil[3]^	2089	1548	1
(4) Tonka, Surinam[4]^	2143	1630	4
(5) Grégoire I^)	3676	1528	8
(6) Grégoire II) F. Guyana[5]*	3697	1437	8
(7) Grégoire III)	3751	1444	8
(8) Barro Colorado, Panama[6]*	2425	1440	2
(9) idem, dry year^	1684	886	1
Africa (▲)			
(1) Tai I, Ivory Coast[7]	2003	1465	1
(2) Tai II, Ivory Coast[7]*	1986	1363	1
(3) Banco I, Ivory Coast[8]^	1800	1145	3
(4) Banco II, Ivory Coast[8]^	1800	1195	2
(5) Yapo, Ivory Coast[8]	1950	1425	1
(6) Guma, Sierra Leone[9]^	5795	1146	3
(7) Yangambi, Congo[10]	1860	1433	1
South-East Asia (□)			
(1) Sungai Tekam C, Malaya[11]	1727	1498	6
(2) Sungai Tekam B[11]^	1781	1606	3
(3) Sungai Lui, Malaya[12,13]^	2410	1516	3
(4) Angat, Philippines[14]^	3236	1232	6
(5) Babinda, Queensland[15,16]*	4037	1421	6
(6) Janlappa, Indonesia[17]	2851	1481	1

[1-17] See footnotes; ^not used in computation of mean ET (see footnotes).

Table 1 continued (footnotes)

[1]Shuttleworth (1988a);
"Tierra firme" forest; lateritic soil; undulating terrain; climate slightly seasonal with 2 mo with 60 < P < 100 mm; Et determined during intensive field campaigns and Ei for almost two years; actual Et used to model stomatal behaviour over time; total ET over 2 yr period synthesised with combined Penman-Monteith-Rutter model using continuous above-canopy climatic data; arguably the best estimate of tropical forest ET available to date.

[2]Leopoldo et al. (1982b);
Environmental setting as above; sandy valley fills; water balance for 130 ha basin; P determined with one gauge; changes in soil water storage (dS) neglected; derived ET too high due to catchment leakage.

[3]Leopoldo et a. (1982a);
Environmental setting as above; water balance for a basin of 2350 ha; P estimated by three gauges near headwater divide; dS not taken into account; ET probably too high.

[4]Poels (1987);
Environmental setting similar to [1-3]; water balance for 295 ha; P determined by one recording and one standard gauge in 80 by 90 m clearing just outside basin; leakage via earthen dam with sharp-crested weir quantified but that via sandy valley bottom assumed negligible; probably overestimate.

[5]Roche (1982);
Climate wetter than for [1-4]; 2 mo with P <60 mm; deep Ultisols on granitic substrate; water balance for basin areas of 320 (G III), 840 (G I) and 1240 ha (G II) ha; G I equipped with 16 rain gauges and considered watertight (Dr. J.M. Fritsch, personal communication); no information available for G II and III but values for ET seem more realistic than for G I.

[6]Dietrich et al. (1982);
Mature semi-deciduous moist tropical forest; relatively thin clayey soils on volcanic agglomerates and siltstones; climate seasonal with 4 mo of P <60 mm; soil water storage capacity ca. 400 mm; water balance for 10-ha basin; P measured with one tipping bucket gauge placed atop roof of field laboratory; weekly gravimetric determinations of topsoil moisture; derived ET may be slightly too high.

[7]Collinet et al. (1984);
Moist semi-deciduous forest; Oxisols on mica schist; climate seasonal; 4 mo of P <100 mm (two dry seasons, one longer than the other); water balance for basins of 120 and 140 ha; no experimental details but basins instrumented and operated by ORSTOM; ET appears realistic.

[8]Huttel (1975);
Moist semi-deciduous forest; sandy (Banco) and clayey (Yapo) soils on sands and schists respectively; Banco II located in valley bottom, others on hill crest; climate similar to [7]; site water balance; P in small clearing; Et derived from dS (neutron probe) down to 230 cm at times of negligible drainage (D); Et at other times evaluated from ET-Turc minus Ei; derived ET approximate; ET-Banco: dry years.

[9]Ledger (1975);
Seasonal (?) evergreen forest on deeply weathered gabbro; high rainfall total; dry season of 5 mo, three of which receive no rain; water balance for a 870 ha basin, 13 per cent occupied by artificial lake; P based on five monthly and one daily rain gauge; streamflow (Q) evaluated from outflows from and changes in storage in reservoir; derived mean annual ET appears reasonable;

[10]Focan & Fripiat (1953);
Secondary (?) seasonal evergreen forest on sandy clay soils; seasonal climate but no details given; site water balance; dS determined twice a day from tensiometers at depths of 15, 35, 55 and 100 cm, from which values for Et and Esoil (and indirectly D) were derived; Ei determined separately but no details given; values for ET from bare soil, grassland and forest appear realistic despite simplicity of approach.

[11]DID (1986);
Dipterocarp rain forest; sandy to clayey Oxisols on lateritic shales (basin C) and clayey Oxisols over andesite (basin A); moderate annual rainfall, uniformly distributed; water balance for basins of 56 ha (C) and 38 ha (A); P determined by two recording gauges (C) or one plus a standard gauge (A); weekly measurement of shallow groundwater levels at two sites per basin; dS neglected; basin A leaky.

[12]Low & Goh (1972); [13]DID (1977);
Dipterocarp rain forest (83 per cent) and rubber plantations (13 per cent) on sandy clay loam soils over granitic rock; equatorial rainall regime but wetter than [11]; water balance for an area of 6810 ha; P estimated with four recording gauges, all below 400 m; areal P likely to be underestimated; possibility of leakage via alluvial fills; derived ET relatively unreliable.

[14]Baconguis (1980);
Dipterocarp rain forest; silt loam (Inceptisol?) over andesitic rock; climate wet with 4 mo with P <60 mm; vegetation water stressed at end of dry season (personal communication S. Baconguis); water balance for a watertight area of 3.8 ha; P via two standard gauges; dS negligible.

[15]Gilmour (1977a); [16]Bonell & Gilmour (1978);
Mesophyll vine rain forest on deep Ultisols with permeability decreasing rapidly with depth; metamorphic rock; high rainfall total with long "dry" (P >100 mm mo^{-1}) season; water balance for South

Creek (25.7 ha); derived ET reliable (all components taken into account).

[17]Calder et al. (1986);
Mature secondary rain forest; deep volcanic soil; equatorial rainfall regime with slightly dry summer; ET computed via Penman-Monteith model with constant values for aerodynamic (r_a) and stomatal (r_s) resistances optimised via measured short-term dS (neutron probe) and long-term Ei (plastic sheets); climatic input data measured at 14 m on ridge in forest; despite simplifying assumptions derived ET feasible.

Some of the higher values quoted in Table 1 (e.g. Ducke Reserve (* 2) and Bacia Modelo (* 3) in Amazonia, also Tonka (* 4) in Surinam) must reflect varying degrees of basin leakage in view of the estimate obtained by micrometeorological methods under very similar conditions by Shuttleworth (1988a). Likewise, values exceeding those for nearby basins (e.g. Grégoire I (* 5), possibly Tai I (▲ 1) and Sungai Tekam (□ 2)) must be treated with caution since, again, catchment leakage cannot be ruled out.

Also, it must be borne in mind that estimates based on site water balances (Yapo (▲ 5), and Yangambi (▲ 7)) suffer from a somewhat poorly quantified drainage term which may have led to an overestimation of ET. Finally, the figure derived for a mature secondary forest in Java (□ 6) represents a computed value involving a number of simplifying assumptions (see footnotes in Table 1).

Clearly, extracting an average value for ET from the data presented in Table 1 remains a rather subjective affair although eventual differences remain small. Values obtained from various combinations range between 1400 mm yr^{-1} (n = 6; based on studies marked with an asterisc which may be considered the most stringent) and 1430 mm (all studies except those marked by ^; n = 11). Nothing meaningful can be stated at present on possible differences between the three major rain forest blocks and more studies, preferably using novel techniques, are desirable, particularly for South-East Asia (cf. the variation quoted in Table 1).

However, the annual ET for a particular forest should not be regarded as static. It tends to be correlated with annual rainfall as a result of increased rainfall interception in wet years and a limiting of transpiration by soil moisture deficits in dry years (Blackie 1979b; Dietrich et al. 1982; Shuttleworth 1988a; see the next two sections).

On physical grounds Calder et al. (1986b) suggested that the evaporation equivalent of the net radiation received by a rain forest canopy that is rarely short of water would provide a good first estimate of ET. Shuttleworth (1988a) reported for an Amazonian rain forest that on fine days typically 75 to 80 per cent of energy was used for transpiration (i.e. evaporation from a dry canopy) but that on rainy days water evaporating from a wet canopy was absorbing energy in excess of that locally available as radiation (see also section 2.3.4). Over a period of two years ET accounted for almost 90 per cent of the net radiant energy measured above the canopy at this slightly seasonal site. Results for well-watered grass and rubber trees (young leaves) in West Africa suggested values of 86 and 80 per cent respectively (Montény 1986).

Annual values of ET in the tropics apparently fall with a decrease in annual rainfall on a broad regional scale (Solomon 1967; Rodier & Vuillaume 1970), but reports on local variations are rare. In his work on several forest types in seasonal western Venezuela, Franco (1979) showed annual ET values from 900 to 1300 mm for deciduous to seasonal evergreen forests. He presented evidence that the water holding capacity of the soils governed both the spatial distribution of forest types and the rate at which ET changed during the dry season (cf. section 2.3.5).

Results for montane forests

The small data set for montane forests contains some of the best long-term records available for forested basins in the tropics. According to the results presented in Figure 9 and Table 2, ET in montane forests is not strongly correlated with altitude or rainfall, mainly because of the rather high values found for some African forests. Apart from the rather extreme value reported by Richardson (1982) for a lower montane forest in Jamaica (attributed by her to high rainfall and breezy conditions but more likely to reflect basin leakage), ET for the other lower montane forests (excluding (semi) "cloud forests" and no's ▲ 9 and 12) converges around a value of 1225 mm yr-1 (range 1155 - 1295, n = 5). The latter is surprisingly close to the mean for lowland forests (ca. 1415 mm; Table 1), especially in view of the large difference in average altitude - about 100 m a.s.l. versus about 1750 m a.s.l. - between the two groups.

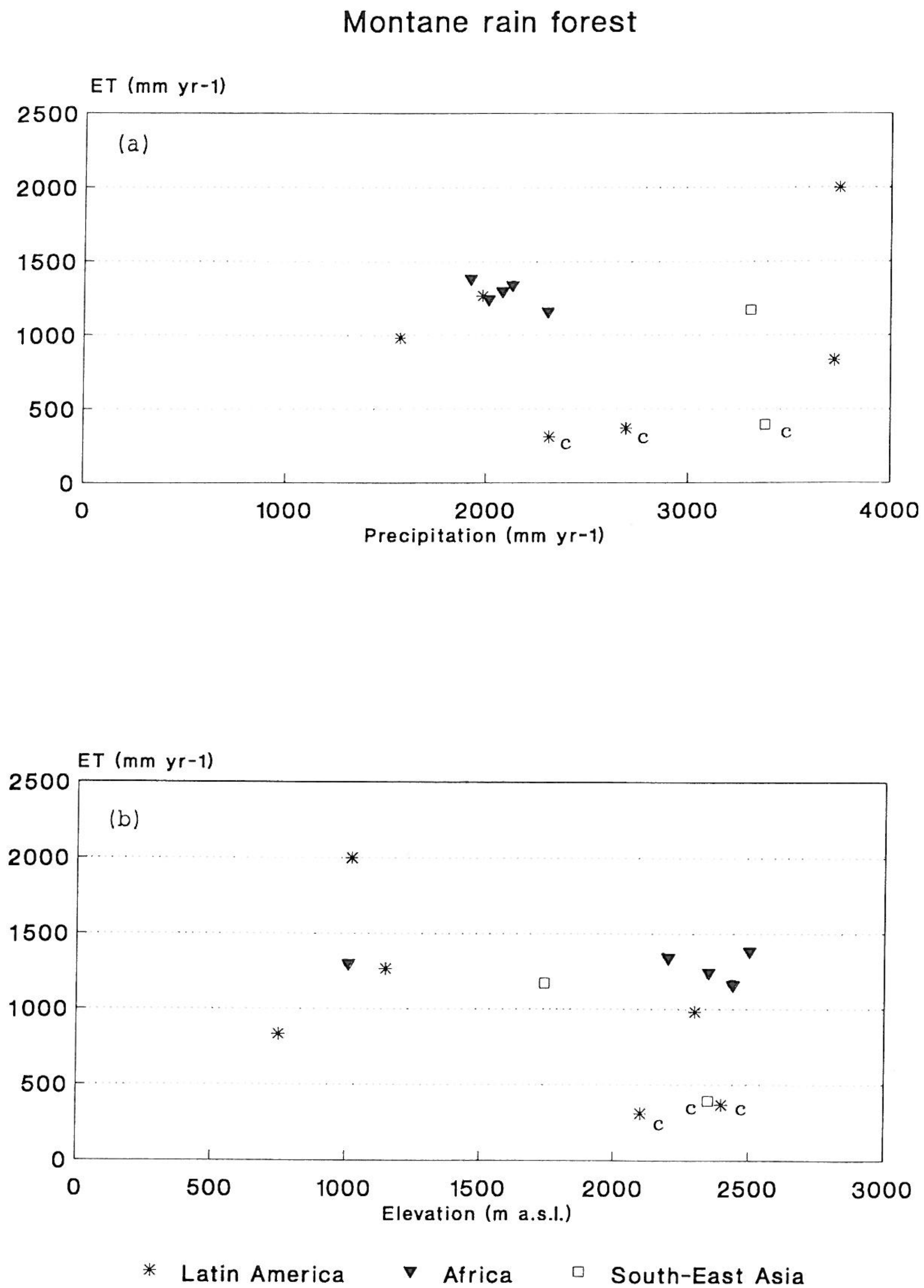

Figure 9. Annual ET versus precipitation (a) and altitude (b) for selected tropical montane forests. c is "cloud forest" (cf. Table 2).

TABLE 2. Evapotranspiration (ET) versus precipitation (P) and elevation (H) for selected tropical montane rain forests

Location	P (mm yr^{-1})	ET (mm yr^{-1})	H (m a.s.l.)
Latin America (*)			
(10) Blue Mnts., Jamaica[1]	3746	1998	1020
(11) Sierra Nevada, Colombia[2,3]	1983	1265	1150
(12) Idem[2,3]	2316	308[c]	2100
(13) San Eusebio, Venezuela[4]	1576	980	2300
(14) Luquillo Mnts., P. Rico[5]	3725	831	750
(15) Rio Macho, Costa Rica[6]	2697	366[c]	2400
Africa (▲)			
(8) Périnet, Madagascar[7]	2081	1295	1010
(9) Kericho, Kenya[8,9]	2130	1337	2200
(10) Idem, subbasin[8,9]	2013	1240	2350
(11) Kimakia, Kenya[10]	2307	1156	2440
(12) Mbeya, Tanzania[11,12]	1924	1381	2500
South-East Asia (□)			
(7) Ciwidey, Indonesia[13]	3306	1170	1740
(8) Mt. Data, Philippines[14]	3382	392[c]	2350

[c] "cloud forest"; [1-14] see footnotes below:

[1] Richardson (1982);
Lower montane rain forest (LMRF); 777-1265 m a.s.l.; gravelly sandy loam (<1 m); porphyrites and conglomerates; high rainfall; water balance for 38.5 ha basin; one recording raingauge at basin outlet; dS measured twice a week over transect (neutron probe); derived ET very high; catchment presumably leaky.

[2] Hermann (1970); [3] Hermann (1971);
LMRF at 1150 m and "cloud forest" at 2100 m; colluvial soils; schists; 4 months with P <60 mm but no water stress; water balance for "small catchments"; no experimental details; occult P not included in figure but estimated at 10 per cent of normal P.

[4] Steinhardt (1979);
Mossy LMRF (transition to "cloud forest"); 2300 m; gley soils;

silty to clayey sedimentary rock; low P with 4 mo with P <60 mm; contribution occult P unknown; ET estimated from energy balance including numerous assumptions and hence approximate.

[5]Frangi & Lugo (1985);
Alluvial palm forest within LMRF at 750 m; gleyed alluvial soils over andesitic rock; wet with P uniformly distributed over year; site water balance (catchment leaking severely); P with above-canopy recording gauge; Ei from 17 throughfall and 26 stemflow collectors; Et from regression between average above-canopy vapour pressure deficit and diurnal rate of groundwater level change during dry days; ET = Et + Ei.

[6]Calvo (1986);
"Cloud forest"; 1960-2840 m; Inceptisols; sandstone and igneous rock; 2 mo with 60< P <100 mm; water balance for 4740 ha basin (watertight); P based on 3 gauges covering range in H; excluding occult P.

[7]Bailly et al. (1974);
Montane seasonal forest; 930-1095 m; sandy to clayey Ultisols; gneiss; 7 mo with P <100 mm (three <60 mm); water balance for 101 ha basin (D4); excellent raingauge network.

[8]Blackie (1979a); [9]Eeles (1979);
MRF; 2200 m; deep Oxisols; phonolitic lava; 2 mo with P just <100 mm; water balance for 544 ha Lagan catchment (watertight) and 186 ha Sambret headwater area (MRF plus bamboo); excellent raingauge network; dS via monthly gravimetric (down to 3.2 m at three sites) and (from 1968 onwards) neutron probe readings (down to 4.5 m; every ten days during dry season)..

[10]Blackie (1979b);
MRF dominated by bamboo; 2440 m; deep soils in pyroclasts over basalt weathered into dense clay; four months of ca. 85 mm; water balance for catchment C (64.9 ha); procedures as described for (9).

[11]Edwards (1979); [12]Blackie & Edwards (1979);
Montane "evergreen" forest (67 %), scrub and grasses (33 %); 1 m volcanic ash over deeply weathered gneiss; 6 mo with P <60 mm (four entirely dry); water balance for catchment C (16.3 ha); excellent raingauge network; dS gravimetric as above.

[13]Gonggrijp (1941b);
LMRF at 1740 m; deep volcanic soils; no month with P <60 mm; water balance for 19.2 ha basin (S.25); P based on four standard gauges; leakage possible in view of spatial variation in Q.

[14]De los Santos (1981);
Mossy MRF ("cloud forest"); 2350 m; sandy loam; basic volcanics; wet with 1-3 mo with P <60 mm; water balance for 7.2 ha basin (Left Fork); P measured with three standard and one recording gauge; occult P not included; large variation in derived ET between two years.

There are no data on net radiation for many of the forests quoted in Table 2, and as such it is difficult to compare average radiant energy in montane and lowland environments. Schmidt (1950) showed that in Java, although sunshine duration decreased strongly with altitude, the effect was lessened by a concurrent increase in radiation intensity. Interestingly, Körner & Cochrane (1985) reported a statistically significant increase in stomatal conductance (potentially leading to enhanced water use) with altitude for eucalypt forests in south-eastern Australia. However, it should be borne in mind that virtually all of the studies quoted in Table 2 are situated in mountainous volcanic terrain and may therefore be prone to subterranean water movements (Gonggrijp 1941b; Rijsdijk & Bruijnzeel 1990). Until independent estimates of ET for tropical montane forests become available, it cannot be ruled out that the above-mentioned average represents an overestimate.

The few examples of "cloud forest"' (no's * 12 and 15, □8 and to a lesser extent * 13 and 14 in Figure 9) exhibit considerably lower values of ET (308-392 mm yr^{-1}) which must be attributed to the combined effects of low transpiration rates under the prevailing climatic conditions (low radiation inputs, high to very high humidity; Gates 1969; Weaver et al. 1973; Cavelier 1988) and to increased precipitation inputs through cloud stripping (Zadroga 1981). As such these low values are apparent only and have to be "corrected" by adding corresponding amounts of occult precipitation. No direct estimates of contributions by occult precipitation to the "cloud forests" of Figure 9 are available, but in two cases such measurements were made in similar forests nearby, viz. by Vis (1986) in Colombia and by Caceres (1981) in Costa Rica. Combining corresponding figures one obtains values for annual ET of 570 and 775 mm for the Colombian and Costa Rican forest respectively, i.e. still considerably below the overall mean of 1225 mm.

Our hydrological knowledge of montane forests in general and of "cloud forests" in particular is still fragmentary, even though they have received considerable attention from plant ecologists and physiologists trying to explain their often peculiar stature (Howard 1968, 1969; Gates 1969; Weaver et al. 1973, 1986; Tanner 1977, 1985; Medina et al. 1981; Tanner & Kapos 1985; Cavelier 1988; Proctor et al. 1988, 1989; Veneklaas 1990; see also summaries by Grubb (1977, 1989) and Whitmore (1989)).

It is of interest to examine the magnitude of the two major components of ET, viz. water vapour transfer from a wet canopy (interception Ei) and from a dry canopy (transpiration Et). The former governs the amount of rainfall reaching the soil, and hence the amount of water available for transpiration.

2.3.4 Rainfall interception

The rainfall intercepted by the forest (Ei) can be derived from measurements of incident rainfall and net precipitation (section 2.1). Not surprisingly in the light of the above remarks on the reliability of many throughfall studies, reported estimates of Ei in tropical forests vary considerably, viz. between 4.5 per cent (Jordan & Heuveldop 1981) and 45 per cent (Read 1977) of gross rainfall. Concentrating on the more reliable studies of throughfall used previously (section 2.3.2) yields average annual values for Ei of 13 (range 4.5 - 22) per cent of rainfall in the case of lowland forests (n = 14) and 18 (range 10 - 24) per cent for montane forests (n = 6; excluding "cloud forests"). In both cases, a figure of 1 per cent was used to account for stemflow (L.A. Bruijnzeel, unpublished manuscript).

The higher value for montane forests might reflect the generally lower rainfall intensities prevailing on tropical mountains (Braak 1921; Riehl 1979), but may be an artefact of the small data set and the choice of the stemflow figure. Measurements of stemflow in lower montane forests are rare and have produced variable results ranging between 1 and 10 per cent of gross rainfall (Steinhardt 1979; Mamanteo & Veracion 1985; Frangi & Lugo 1985). Annual Ei for "mossy" or "cloud forest" ranges from about 10 per cent (Caceres 1981; Mamanteo & Veracion 1985; Vis 1986; Veneklaas 1990) to negative values due to cloud stripping on particularly exposed locations (Weaver 1972; Mamanteo & Veracion 1985).

Again it should be remembered that the above-mentioned average values of Ei are probably overestimates as the throughfall figures on which they are based must be considered as conservative (section 2.3.2).

Expressing values of Ei as a fraction of incident precipitation may be useful in comparing results from different locations, it does not give direct information on actual rates of evaporation from a wet forest canopy (Ewet). The latter, but especially the amount of water remaining on the foliage at the end of a storm, constitutes the bulk of the interception loss under humid tropical conditions (Bruijnzeel & Wiersum 1987; Lloyd et al. 1988; Hutjes et al. 1990). Values for Ewet in rain forested areas are about 0.2 mm hr^{-1} (Murdiyarso 1985; Lloyd et al. 1988; Hutjes et al. 1990) but higher values have been derived for tropical plantations of limited areal extent and surrounded by shorter vegetation (Rao 1987; Bruijnzeel & Wiersum 1987).

As indicated in the discussion of total forest ET, Ewet often exceeds available amounts of net radiation (Shuttleworth 1988) and an additional supply of energy is therefore required. The wet forest canopy is thought to act as a sink for advected energy derived from the heat content of (slightly warmer) air passing overhead (Rutter 1967; Stewart 1977). This process is capable of producing large absolute interception losses during heavy storms (say, larger than 100 mm) (Bruijnzeel & Wiersum 1987; Scatena 1990) and overrides effects of structural differences between vegetative covers (cf. Table 5 in Bruijnzeel 1988; Scatena 1990).

The fact that convectional rainfall in the tropics generally exhibits extreme spatial variation (Riehl 1979) makes it plausible that the enhanced evaporation during and immediately after rain at one site is, at least partly, supported by energy brought into the atmosphere as sensible heat at other sites where the canopy remained dry (Shuttleworth 1988a). Also, under more maritime conditions, there may be considerable transfer of heat from the ocean to the atmoshere, capable of maintaining high evaporation rates at times when radiative inputs are low, winds feeble and humidities high (Pearce et al. 1980; Bruijnzeel & Wiersum 1987).

2.3.5 Transpiration

In contrast to the many studies of Ei, there are few accounts of evaporation from a dry canopy (i.e. transpiration, Et) for moist tropical forests. Several techniques can be used to estimate Et (rates), viz. hydrological, micro-meteorological, plant physiological and "empirical" ones (Shuttleworth 1979).

The hydrological technique basically combines data on ET (usually obtained from a plot or catchment water balance) with information on Ei, finding Et by difference (for weekly or longer periods) and neglecting evaporation from the forest floor (e.g. Leopoldo et al. 1982a, b). The estimate must be regarded as potentially crude due to the possibility of accumulating errors and basin leakage (cf. Lee 1970; Lloyd & Marques-Filho 1988). Thus values of annual Et for two similar forests in Amazonia obtained by this method differed by almost 300 mm (987 vs. 1280 mm; Leopoldo et al. 1982a, b). As we shall see, most estimates have been made in this way and should be interpreted with care.

A potentially more reliable variant of the hydrological technique involves the monitoring of changes in soil water during (extended) periods of low rainfall (Huttel 1975; Cooper 1979; Eeles 1979; Franco 1979; Calder et al. 1986b). Such measurements should be carried out to a depth of at least 300 cm if the soil is as deep or deeper than that, as trees are known to extract water from depths of at least 150 cm in lowland rain forests (Huttel 1975; Guehl 1983; Calder et al. 1986b) and occasionally (e.g. during droughts) down to a depth of about 5 m (Eeles 1979; Poels 1987). Naturally, appropriate attention should be paid to problems of spatial variation if meaningful results are to be obtained (Rambal et al. 1984).

Yet another variant of the technique which has proved particularly useful in more seasonal climates is the so-called zero flux plane method which involves concurrent measurements of soil water content and soil water potential with depth and enables one to elegantly separate soil moisture depletion due to Et and due to drainage (Cooper 1979).

The repeated measurement of soil moisture content throughout the root zone and at an adequate number of points generally precludes the use of destructive sampling techniques (Reynolds 1970). Up to now most investigations in tropical forests have used neutron probe equipment (Huttel 1975; Eeles 1979; Calder et al. 1986b), which enables the sampling of a large number of points in a relatively short time without site disturbance. Interestingly, a non-nuclear device which is superior to the neutron probe in several respects, the capacitance probe, has become commercially available recently (Dean et al. 1987; Bell et al. 1987). Recent tests of the instrument in pine plantations on Viti Levu, Fiji, were promising indeed (M.J. Waterloo and L.A. Bruijnzeel, unpublished data).

Relevant data on soil moisture depletion were collected in lowland rain forests in Ivory Coast by Huttel (1975) using a neutron probe. Maximum rates of depletion occurred at the onset and end of the rainy season, whilst minima were observed at the height of the dry season (cf. Franco 1979; Institute of Hydrology 1990). Huttel's data were published in a graphical form only and an opportunity has been missed to construct a mathematical model describing the decline in rates of Et as a function of soil moisture for these forests (cf. Eeles 1979).

Micro-meteorological methods have yielded some of the best data available on tropical forest Et (Shuttleworth 1988a) and in view of advances made in recent years in the reliability, portability and versatility of equipment and peripherals, it is to be expected that considerable progress can be made in this respect using such techniques at a small number of carefully selected research sites.

Several plant physiological techniques, such as the injection and monitoring of isotope tracers, such as tritium (Kline et al. 1970; Jordan & Kline 1977) or deuterium (Calder et al. 1986a; Institute of Hydrology 1990), the heat pulse method (Kenworthy 1969; Yoshikawa et al. 1986) as well as potometry (Coster 1937; Wanner & Soerohaldoko 1979), have also been used to gain an idea of rates of Et in tropical forests. Since such measurements are made on individual trees the problem of extrapolation to the forest as a whole remains (Denmead 1984).

Jordan & Kline (1977) reported a high correlation between transpiration volumes (l day^{-1}) per tree as determined by isotope tracing and sapwood cross sectional area for a set of twenty different trees in a lowland rain forest in the Amazon territory of Venezuela. Combination with the results from a tree diameter inventory yielded an estimate of forest Et of 2186 mm yr-1 (Jordan & Heuveldop 1981), but this value is clearly an overestimation (cf. Figure 8). Nevertheless, the potential of the isotope injection technique was demonstrated by Waring & Roberts (1979) and Waring et al. (1980) who found good agreement between rates of water uptake as determined by isotope tracers on the one hand and whole-tree potometry and the Penman Monteith transpiration model (see below) on the other for *Pinus sylvestris* in Scotland. Similarly, estimates of Et for eucalypt trees in south-western India based on deuterium tracing compared favourably with rates of water consumption as determined by whole-tree potometry (Institute of Hydrology 1990).

Extrapolation techniques for the heat pulse technique are basically the same as those for isotope tracing methods. From the data on tree diameter and sapflow rates reported for a Sumatran rain forest by Yoshikawa et al. (1986) a highly significant linear regression equation could be computed between sap flow rates (cm3 hr-1) and total cross sectional area (Bruijnzeel, unpublished manuscript). However, trees of the same species and of similar age and size may transpire at quite different rates (Ladefoged 1963; Doley & Grieve 1966; Cermak & Kucera 1987; Werk et al. 1988; Hatton & Vertessy 1989) and hence the sample sizes necessary to characterise overall forest water use can be quite large (Denmead 1984). The recent development of a heat pulse logger system by CSIRO capable of simultaneously monitoring the sapflow of up to 16 trees (Durham & Hatton 1989) constitutes a most interesting development in this respect. Further work is desirable and should be backed up by independent estimates of Et.

Lastly, a more "empirical" approach is the use of the physically-based combination formula developed by Penman (1948, 1956) as extended by Monteith (1965) to describe evaporation from a vegetated surface. The Penman Monteith model proved to be a powerful tool in the prediction of evaporation from both wet and dry canopies in the temperate zone (Calder 1977; Gash & Morton 1977) and recently it has been shown to be capable of producing plausible results for tropical forests as well (Whitehead et al. 1981; Calder et al. 1986b; Shuttleworth 1988a; Roberts et al. 1990). A major difficulty with the application of the Penman Monteith formula, however, is associated with the "canopy resistance" (r_c) to water vapour transfer, which is not only difficult to estimate for a species-rich forest, but is also a strong determinant of the formula's output (Beven 1979).

Roberts et al. (1990) made extensive measurements of stomatal resistance throughout the canopy of a rain forest near Manaus, Brazil. They found the stomatal resistance of leaves in the upper half of the canopy to decrease with increasing solar radiation, whilst a positive correlation existed with atmospheric humidity deficit. Seasonal differences in resistance were ascribed to effects of soil moisture deficit and seasonal differences in leaf age. When Et was calculated for various strata in the canopy by inserting measured stomatal resistances in the Penman Monteith formula, the results compared closely with independent estimates by a micro-meteorological technique (Institute of Hydrology 1988).

However, converting leaf stomatal resistances to overall canopy resistance requires knowledge of a forest's leaf area index (LAI) and of its distribution in the vertical (Dolman 1987). Since information on LAI in tropical forests is still very limited, this is likely to remain the bottleneck for the use of the Penman-Monteith formula. Luvall (1984) estimated LAI for a forest in Costa Rica by counting the number of contacts of several lines dropped through the canopy whilst Lang et al. (1985) and Pierce & Running (1988) suggested the use of the light transmitting characteristics of a canopy. Alternatively, if Et is known from independent measurements, r_c can be evaluated by solving the Penamn Monteith equation (Shuttleworth 1988a). This information may then be used in future computations with the P-M equation for similar forest types.

Most of the studies on lowland forest ET (Figure 8) have also reported estimates of Ei which were combined to compute approximate annual values for Et. This resulted in estimates ranging from 885 to 1285 mm yr^{-1} with an average of 1045 mm (n = 9). The latter value comes close to the 980 mm obtained for an Amazonian forest via micrometeorological techniques as reported by Shuttleworth (1988a).

Transpiration for markedly seasonal forests in Figure 8 (*6, Panama, .3, Queensland) was below 500 mm yr-1. This low value may partly be caused by the application of relatively high interception values (Read (1977) and Gilmour (1975) respectively), but Franco (1979) observed a similarly low value (630 mm) in semi-deciduous forest in Western Venezuela. Values of Et for montane forests (excluding "cloud forests") varied between 510 and 830 mm yr-1 with no relation to elevation. The data set, however, is small (n = 5).

For "cloud forests", the available information is limited to instantaneous estimates via physiological techniques for forests in Puerto Rico (Gates 1969; Weaver et al. 1973), which in combination with a leaf area index figure of 2 (reported by Weaver et al. (1986)) and extrapolated to a year, suggested a value for Et of less than 75 mm yr-1 (0.2 mm day-1), a very low value indeed. Alternatively one can gain an idea of "cloud forest" Et by combining the data on ET and Ei for Colombian and Costa Rican forests collected by Hermann (1970) and Vis (1986) and by Calvo (1986) and Caceres (1981) respectively. Correcting for occult precipitation inputs (ca. 10 per cent), annual values of about 285 mm (0.7 mm day-1) for the Colombian forest and about 510 mm (1.4 mm day-1) for the Costa Rican forest were obtained. A similar exercise for a "cloud forest" in the northern part of the Philippines (based on rainfall and streamflow data presented by De los Santos (1981) and interception data reported in Mamanteo & Veracion 1985) yielded an estimate of Et of about 350 mm yr^{-1}. Such low values are probably caused by the weather conditions in these forests (Baynton 1968; Weaver et al. 1973), rather than by high stomatal resistances (Cintron 1970; Kapos & Tanner 1985). Also in view of the fact that water stress rarely, if at all, occurs in "cloud forest" trees (Hermann 1971; Kapos & Tanner 1985), their low stomatal resistance may be adaptative in that it may enable the trees to rapidly increase transpiration rates whenever weather conditions permit. Several workers (reviewed by Leigh 1975) suggested that persistent fog might cause forest stunting by blocking transpiration and, consequently, nutrient uptake. However, Grubb (1977) provided arguments against this idea and suggested that, if nutrient uptake is inhibited by cloudy conditions, it is much more likely to be due to low light intensities and reduced supply of metabolites to the roots. Similarly, leaves of undergrowth species in forests in Indonesia and Hawaii were shown to have low evaporative resistance by Wanner & Soerohaldoko (1979) and Pearcy & Calkin (1983) respectively, a finding they interpreted as an adaptation to sporadic exposure to sunlight.

Summarising, our knowledge of transpiration rates in lowland tropical forests has increased substantially in the last few years but more work is needed, especially on physiological controls and in montane forests.

Now that the overall hydrological framework for moist tropical forests has been described, the forest nutrient cycle and the capacity to regulate nutrient losses that is so often ascribed to these forests will be examined in the next chapter.

3 NUTRIENT INPUT - OUTPUT BUDGETS

3.1 Introduction

Although the classical view of a tropical rain forest as having a relatively rich nutrient economy perched on a nutrient poor substrate (Whittaker 1975) has been shown by Proctor (1987) to be an overgeneralization, the proportional extent of soils with moderate to low fertility in the moist lowland tropics nevertheless is far greater (ca. 77 per cent) than that occupied by relatively fertile soils (Vitousek & Sanford 1986).

Since the ratio of solid matter (soil and weathered rock) over the liquid phase (percolating water) is very large, small changes in the chemical composition of the former may show up as quite detectable changes in the chemistry of the latter (Verstraten 1980). As such, the amounts of nutrients carried by streams can be regarded as a useful, be it relative, indicator of an area's nutrient status, both before and after a change in land use (Likens et al. 1977).

Macro-nutrient input-output budgets for a limited number of tropical forest ecosystems have been listed and to some extent discussed in Vitousek & Sanford (1986) and Proctor (1987) but an in-depth analysis of the available information on tropical forest nutrient budgets has been lacking up to now. The present chapter is an attempt to fill this gap. In addition, it is intended to serve as a point of reference for the evaluation of hydrochemical responses brought about by the various kinds of forest disturbance presented in Chapter 5.

After a brief introductory description of the forest nutrient cycle and a discussion of methodological problems related to the computation of forest nutrient budgets in general and in the tropics in particular, the results of some twenty five input-output studies in (sub)tropical forests are presented and evaluated.

3.2 The forest nutrient cycle

Nutrient cycling in forests involves a complex set of direct and indirect feedback mechanisms between soil and vegetation. Obviously, a detailed description of these interactions is outside the scope of the present paper (cf. Likens et al. 1977; Coleman et al. 1983; Medina 1984).

Briefly, a forest ecosystem constitutes an open system with chemical elements entering and leaving, thereby linking the system to the larger global cycles. In addition, there is the tendency exhibited by a number of elements to cycle continuously within the ecosystem (Figure 10a). Sometimes the term "biogeochemical cycling" is used to denote this phenomenon as elements tend to move from non-living components ("geo") to living organisms ("bio") and back (Odum 1971). The forest biogeochemical cycle can be conveniently thought of as transfers of nutrients between a number of compartments or pools (Likens et al. 1977; Figure 10b).

Proctor (1987) provided the following concise description of the major pathways linking the various storage pools:

> "Nutrients enter the ecosystem with the rain, deposition of dust and aerosols, (in the case of nitrogen) by fixation by micro-organisms above and below ground, and (except for nitrogen) by weathering of the underlying rock. The major above-ground pool of nutrients is the canopy (defined as the total plant community) and there is a flow of nutrients from this to the forest floor in small and large litterfall and in throughfall and stemflow of rainwater, which usually becomes enriched by nutrients from leaves and bark. A proportion of the above- ground nutrients is in dead organic matter such as standing dead trees and small and large litter lying on the forest floor. Nutrients are gradually released from the dead matter by decomposition mediated by soil animals and micro-organisms. Decomposition is complex and can involve immobilization of nutrients as well as their release. Nutrients are taken up from the exchange complexes of the soil by roots (probably usually in association with mycorrhizal fungi) which provide a living below-ground pool and which export them to the canopy. The roots release nutrients to the soil as secretions and by the death and decomposition of their parts. Permanent loss of nutrients occurs through surface erosion, fires, loss in drainage water and in the case of nitrogen by abiotic or microbial denitrific ation. Some, particularly phosphorus, may effectively leave the system by conversion into insoluble inorganic forms within the soil".

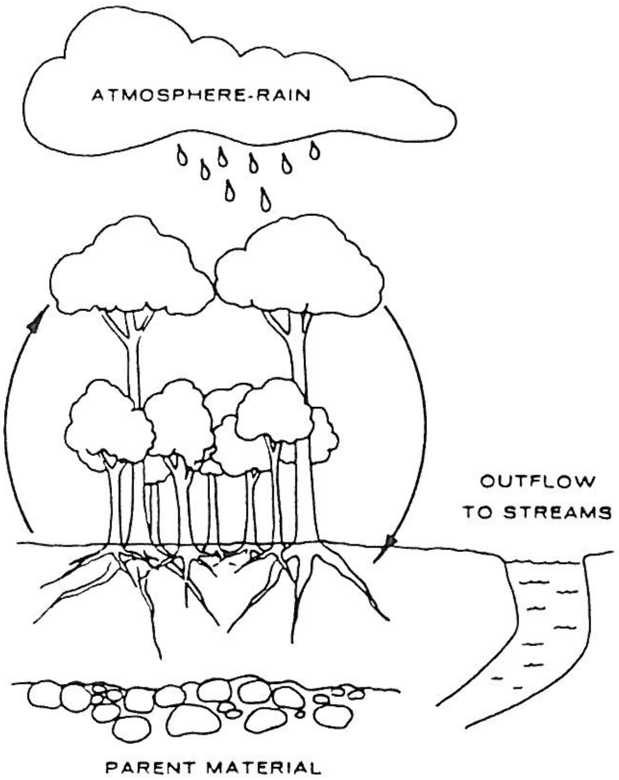

Figure 10a. Simplified diagram of cycling within the active ecological community (after Golley et al. 1975).

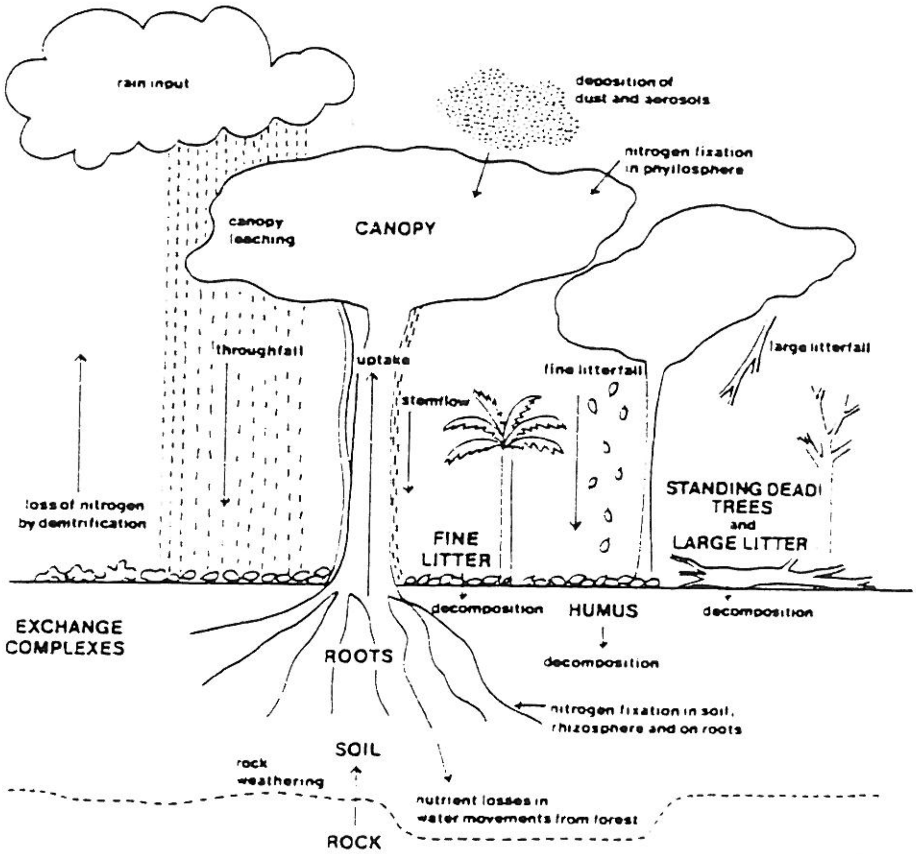

Figure 10b. Nutrient cycling pathways in tropical forest (after Proctor 1987).

Tropical forests growing on nutrient poor substrates are only able to maintain themselves at a high biomass level through an array of nutrient conserving mechanisms, producing a relatively "tight" or "closed" nutrient cycle with only small amounts of nutrients leaking from the system (Herrera et al. 1978; Brinkmann 1985). Forests growing on more fertile substrates, on the other hand, will exhibit a more "open" type of nutrient cycle (Baillie 1989).

It is important to make a distinction between primary and old secondary forests, which have essentially attained a state of dynamic equilibrium (Uhl 1982; Lieberman et al. 1985), and forest plantations or young secondary forests, which generally represent rapidly aggrading systems. In the former, the rate of nutrient uptake needed to maintain forest biomass at a certain level will be covered more or less by the amounts of nutrients released from decomposing litter and logs, supplemented by contributions via wet and dry deposition (Bruijnzeel 1989a). Forest plantations and young secondary forests in the tropics, however, usually incorporate considerable amounts of nutrients in their biomass in a relatively short period of time (Russell 1983; Bruijnzeel & Wiersum 1985; Jaffré 1985). Net uptake rates normally exceed the amounts supplied by litterfall and wash from the canopy, especially during the first decade (Golley et al. 1975; Ewel 1976; Egunjobi & Bada 1979; Bruijnzeel 1983a; Gholz et al. 1985). The remainder will have to be supplied by the soil and this, as we shall see in Chapter 5, may present difficulties under certain conditions (Russell 1983; Bruijnzeel & Wiersum 1985).

Whereas in the short run nutrient availability is regulated by the balance between processes releasing nutrients into available forms and those removing them (Figure 10b), in the long run the nutrient status of a forest ecosystem depends on the balance between nutrient inputs (wet and dry deposition, mineral weathering, gas absorption/fixation) and outputs (mainly leaching and volatilization; Vitousek & Sanford 1986; Lewis et al. 1987; Figure 10a,b).

3.3 Methodological aspects

3.3.1 Computation of nutrient losses

Nutrient losses reported in the literature have been obtained by a vare 3 continued (footnotes)

rera (1979); [2]Buschbacher (1984)
aatinga" forest; highly leached and frequently water-logged sands; "black ater" area; amount of drainage computed by subtracting estimated evaporation total for Rio Negro river basin (126760 km^2) from average rainfall at San Carlos; nutrient concentrations of drainage water approximated by [a]weekly collections of 15 zero-tension lysimeters at -12 cm during October and November 1975, [b]groundwater at -100 cm (no specification of time period) and [c]streamwater (August 1975 - December 1976); nitrogen comprises inorganic fractions only; rainfall sampled between November 1975 and December 1976; [d]difference between atmospheric inputs as estimated by Buschbacher (1984) and streamwater outputs as before.

[3]Lewis (1986); [4]Lewis et al. (1987)
Undisturbed lowland forest; highly depleted soils on Precambrian rocks; spodosols in depressions; "black water" river; catchment area 47,500 km^2; bi-weekly depth integrated sampling of river water at four points across the stream between May 1982 and April 1984; nutrient inputs estimated via bulk precipitation inputs at basin outlet and adjusted on the basis of basin equilibrium for sulphur and chlorine in/outputs; dry-season contributions not included; high standards for storage and analysis of water samples.

[5]Jordan (1982); [6]Jordan & Heuveldop (1981); [7]Jordan et al. (1982); [2]Buschbacher (1984)
"Tierra firme" forest; highly leached oxisols; amount of drainage computed by subtracting "class A" pan evaporation from rainfall; average nutrient concentrations in drainage water as collected by 24 zero-tension lysimeters was computed by weighting samples obtained from lysimeters at -12 cm four times as heavily as concentrations at -40 cm to account for subsurface lateral flow; precipitation sampled from 20 polypropylene bottles with their orifices covered by a thin layer of glass wool; quoted data concern 1976-1979 annual totals; phosphorus as phosphate only; nitrogen comprises inorganic fractions; [e]inputs from Buschbacher (1984); outputs computed by subtracting average lowland rain forest ET (1425 mm yr^{-1}, Bruijnzeel 1989b) from rainfall to obtain drainage rate multiplied with average concentrations for lysimeters at -17 and -30 cm in a depleted ultisol as derived from unpublished data over 2.5 year by Buschbacher in Medina & Cuevas (1989).

[8]Lam (1978)
Dense natural regrowth of Pinus massoniana; deeply weathered granite; red-yellow podzolic soils; catchment area 24 ha; observation period July 1971 - October 1972 and wetter than usual; vegetation still acquiring biomass; all values approximate as they were derived from a graph and converted to annual values by multiplication with 0.8.

[9]Russell (1983)
Lowland rain forest; sandy ultisol low in calcium, potassium and phosphorus; drainage approximated as 50% of rainfall (Molion, 1975); corresponding nutrient concentrations by sampling six (?) zero-tension lysimeters at -30 cm during three periods (9 samples) in 1981; rainfall sampled daily at five stations between January 1981 and January 1982 using the approach of Jordan (1982) at San Carlos, Venezuela.

[10]Roose (1981)
Old secondary and slightly disturbed "evergreen" forest; Tertiary sandy and clayey deposits; oxisols; drainage estimated by a modified Thornthwaite soil water budget method (1964-1975); outflow from cylindrical (diameter 63 cm, height up to 120 cm) soil monoliths frequently sampled for chemical analysis and concentrations averaged for surficial and deeper layers; [f]outflow obtained by multiplying average spring water composition with average drainage rate.

[11]Kenworthy (1971)
Partly disturbed Dipterocarp forest; granite; oxisols; catchment area 31 ha; observation period August 1968 - February 1969 with measurements extrapolated to a one-year period; no details given for collection, storage, and analytical procedures; nutrient concentrations in rainwater rather high (Manokaran 1980), suggesting sample enrichment cannot be ruled out; streamflow probably underestimated, possibly because of leaky conditions.

[12]Huttel (1975); [13]Bernhard-Reversat (1975)
Environment as described for [10]; drainage computed via soil water balance with variations in soil water storage measured by neutron probe technique; during wet periods (net downward flow) drainage derived by subtracting forest ET (assumed equal to that computed with the Turc formula) from rainfall; observation period (1969-1971) received below average rainfall totals; two zero-tension lysimeters at -40 cm sampled weekly and analyzed after pooling to monthly samples; rainfall chemistry (1970-1972) approximated by that recorded at nearby Adiopodoumé; [g]nutrient outflow obtained by multiplying average streamwater composition with average drainage rate.

[14]Poels (1987)
"High dryland forest" on slopes and ridges, "marsh/swamp" forest in depressions; sandy loam deposits on granitic (?) Precambrian basement; well-drained yellowish brown oxisols on slopes/ridges, poorly drained grey sands (tropaquents) in valleys; catchment area 140 ha; observation period November 1979 - April 1984; weekly sampling of rainfall and streamflow; rainfall sampled from groundlevel gauge; samples with electrical conductivity > 50 microS/cm discarded; numerous other corrections applied to account for rather weak analytical facilities in Surinam; [h]inorganic fractions based on limited number of samples.

[15]Brinkmann (1983); [16]Brinkmann (1985)
"Terra firme" rainforest with "riverine" forest along streams; Tertiary sedimentary rock, depleted yellow oxisols on plateau and slopes; sandy hydromorphic soils in valleys; several small catchments and plots; observation period 1968-1972.

[17]Franken & Leopoldo (1984)
Vegetation and soils as [15,16]; catchment area 1.3 km^2; observation period September 1976 to September 1977 and rather dry; rainfall and streamflow sampled weekly; concentrations of calcium and magnesium generally within methodological error; [i]phosphate-phosphorus; [j]ammoniacal nitrogen.

Table 3 continued

Location	Annual rainfall (mm)	Annual runoff (mm)	Calcium I	L	I-L
				(kg ha^{-1} yr^{-1})	
			Soils of moderate to high fertility		
(III) Inceptisols/Mollisols/Vertisols					
(13) Watubelah, Indonesia[21-23]*	4670	3590	9.9	29.0	-19.1
Alternate computation[q]				86.3	-76.4
(14) La Selva, Costa Rica[24]+	3675[r]	1250[r]	2.7	5.7	- 3.0
Alternate computation[u]*	4007	2580	3.1	9.2	- 6.1
(15) Kinta Valley, Malaysia[26,25]*	2845	1605	11.4	795	-784
				651[v]	-640[v]
(16) Ei Creek, Papua[27]*	2700	1480	0[w]	24.8	-24.8
(17) Gua Anak Takun, Malaysia[25,26]+*	2440	1255	36.1[x]	764	-728
				583[v]	-547
(18) Lien-Hua-Chi, Taiwan[28]*	2375	840	3.5	20.3	-16.8
(19) Darien, Panama[29]*	1935	855	29	163	-134
			MONTANE FORESTS		
Inceptisols/Ultisols					
(20) Rio Espiritu Santo, Puerto Rico[30-33]*	4550	4260	23	166	-143
(21) Mt. Kerigomna, Papua[34,35]*	3800	3160	3.6	(63)	(-59)
(22) El Verde, Puerto Rico[36,37]*	3920[A]	2150[A]	21.8	43.1	-21.3
(23) Pi-Lu-Chi, Taiwan[38,39]*	2420	1250			
Watershed PL-11		1100	17.8	308	-290
Watershed PL-12		1400	17.6	408	-390
(24) Chiengmai, Thailand[40]*	2035	930	16.0	4.0	+12.0
Ibidem applying 1969 nutrient concentrations in rainfall[41]			5.1	4.0	+1.1
(25) San Eusebio, Venezuela[42]+	1500	565	5.6	1.7[D]	+3.9

*catchment based study
[a-m]see footnotes next pages
+(zero-)tension lysimetry
°total soluble amounts unless indicated otherwise
▲excluding fixation inputs and denitrification losses

Magnesium			Potassium			Phosphorus			Nitrogen		
I	L	I-L	I	L	I-L	I	L°	I-L	I	L	I-L▲
				($kg\ ha^{-1}\ yr^{-1}$)							
4.0	30.5	-26.5	9.6	22.0	-12.4	1.2[n]	0.7[n]	+0.5[n]	15.4[p]	10.6[p]	+4.8[p]
	43.1	-39.1		49.0	-39.4		5.9	-4.7		38	-23
2.8	8.5	-5.7	2.0	3.6	1.6	0.08[s]	0.03[s]	-0.01[s]	1.1[t]	19.4[t]	-18.3[+]
2.6	5.0	-2.4	5.4	21.6	-16.2	0.17[s]	0.03[s]	+0.14[s]	1.7[t]	-5.6[t]	-3.9[t]
1.4	90	-89	3.4	76	-72						
	78	-77[v]	13.5[v]		-10[v]						
0.3	51	-50.7	0.8	14.9	-14.1						
3.4[x?]	45	-42	3.7	20	-16						
	17[v]	-13		3.4[v]	+0.3						
1.2	10.3	-9.1	4.6	12.1	-7.5	0.2[s]	tr.[s]	+0.2[s]	7.7[t]	7.4[t]	+0.3[t]
5	44	-39	9.5	9.3	+0.2	1	0.7	+0.3			
22	72	-50	21	18	+3		0.7[y]		1[z]	2.3[z]	-1.3[z]
1.3	(25)	(-24)	7.3	(22)	(-15)	0.53			6.5		
4.9	15	-10.1	18.2	20.8	-2.6	0.9	1.3	-0.4	14[B]	29[B]	-15[B]
3.9	166	-162	7.3	10.0	-2.7		tr.		9.7[C]	1.6[C]	+8.1[C]
3.9	220	-216	7.2	10.4	-3.2		tr.		9.6[C]	1.6[C]	+8.0[C]
16.1	2.5	+13.6	12.3	2.6	+9.7						
0.2	2.5	-2.3	0.6	2.6	-2.0						
5.2	0.6[D]	+4.6	2.6	2.2[D]	+0.4	1.1[E]	0.3[D]	+0.8	9.9	4.4[D]	+5.5

Table 3 continued (footnotes)

[18]Abdul Rahim & Zulkifli (1986); [19]Zulkifli (1989); [20]Zulkifli et al. (1989)
Undisturbed Dipterocarp forest; deeply weathered granite; deep ultisols (2/3 rd) and oxisols (1/3 rd) of sandy clay (loam) texture; catchment area 29.6 ha; observation period for streamflow quantity and quality (July 1980 - June 1983) included a wet and a dry year with overall precipitation total close to long-term average; rainfall chemistry studied between September 1986 and August 1987 (normal rainfall); possibility of sample enrichment for calcium and magnesium (dust) cannot be ruled out; [k]phosphate phosphorus; [l]inorganic forms only, total nitrogen input 11.4 kg $ha^{-1}yr^{-1}$; [m]applying a runoff figure of 575 mm yr^{-1} to account for deep leakage and using concentrations of calcium, magnesium and potassium in rainfall as determined at nearby Pasoh (Manokaran 1980).

[21]Bruijnzeel (1983a); [22]Bruijnzeel (1983c); [23]Bruijnzeel (1984)
Plantation forest of <u>Agathis dammara</u> (11-35 years old); andesitic tuffs underlain by andesitic breccias; rather fertile andept; catchment area 18.7 ha; observation period December 1976 - February 1978, including severe dry spell; weekly sampling; outflow data normalized for average runoff total; [n]phosphate-phosphorus and based on limited data; pbased on limited number of samples taken in wet season; [q]values corrected for incorporation in aggrading biomass (see Bruijnzeel (1983a) for details) not necessarily representative of outputs for ecosystem in steady state.

[24]Parker (1985)
"Tropical wet" forest; Quaternary basaltic volcanic deposits; "ultic" Andept; drainage rates approximated by site water balance technique using Penman-Monteith evaporation model; [r]observation period 404 days (March 1983 - May 1984) with below-average rainfall; quoted drainage rate would correspond with unrealistically high value for forest ET (2190 mm yr^{-1}); concentration of percolating water obtained from "numerous" suction lysimeters at 70 cm with suction maintained at -300 mbar and sampled every two weeks; rainfall often sampled on an event basis; high standards for collection and analysis of samples; [s]phosphate-phosphorus; [t]inorganic forms; [u]average annual nutrient input as given by Parker (1985); output based on streamwater chemistry by Parker (1985) combined with runoff estimated as rainfall minus average lowland rainforest ET of 1425 mm yr^{-1} (Bruijnzeel 1989b).

[25]Crowther (1987a); [26]Crowther (1987b)
Lowland rainforest; limestone; 40-70 cm deep Rendolls (?) on footslopes with alluvial admixture; amounts of drainage estimated by subtracting forest ET (approximated as 0.83% of "class A" pan evaporation at the nearest meteorological station) from rainfall; nutrient concentrations in drainage water obtained by 3-weekly sampling of one zero-tension lysimeter inserted at contact with bedrock and also [v]by 3 to 6-weekly sampling of large numbers of seepage points in caves for one year; drainage rates considered slight underestimates by the investigator himself; incident precipitation sampled "regularly"; no further details given on collecting procedures.

[27]Turvey (1974)
Colline rainforest, basaltic volcanic agglomerates underlain by Mg-bearing phyllites; "acid red to brown clay soils"; catchment area 16.25 km^2; observation period July 1972 until May 1973 with numerous gaps in records; rainfall (after December 1972) and streamflow sampled at least weekly; [w]nutrient concentrations in rainfall low and at least for calcium considered anomalous (analytical error) by investigator himself.

[25,26]Crowther (1987a,b)
Environmental and procedural details as given for Gua Anak Takun study (17); drainage of soil water sampled by two lysimeters; [x]enriched by dust from quarry.

[28]Horng et al. (1985)
Warm temperate rain forest (evergreen); sandstone and shales; inceptisols of fine silt loam texture; catchment area 3.4 ha; observation period 1984; rainfall sampled per event, streamflow at least daily; catchment probably leaky as nearby catchments exhibited consistently higher runoff totals (ca. 300 mm yr^{-1}).

[29]Golley et al. (1975)
"Evergreen seasonal" forest; shale interbedded with dolomite and calcareous sandstone; black clay soils (Vertisol) with high base saturation; an approximate nutrient budget was computed for the 259 km^2 Sabana catchment over 1967: rainfall data from various stations in the region were combined to give annual total of 1933 mm; rainfall at one location sampled on seven occasions in September 1967, possibly by using a plastic sheet; discharge data available between May and December and extended to annual value by comparison with other catchment; chemical composition of river water is average of a number of streams in the region and based on a very limited number of samples; concentrations for Rio Sabana itself far higher than regional average.

[30]Jordan et al. (1972); [31]Clements & Colon (1975); [32]Brown <u>et al</u>. (1983); [33]Lugo (1986) Lower Montane Rainforest (500-1000 m) grading into dwarf forest at highest elevations; andesitic rocks; generally acid ultisols of clay loam texture and rather low base saturation; catchment area not specified but estimated at ca. 10 km^2 from general maps in [32]; approximate nutrient budget computed by present writer using average rainfall and streamflow data from [33]; streamwater quality data from USGS quoted in [33] (numerous samples over several years); precipitation chemistry: average concentrations for El Verde at 500 m from [30] and [31] (two separate full years) and for dwarf forest at 1000 m (Trinidad Pizarro (1985) and personal communication in [33]) weighted by elevation zones as given in [33]; upper 10% of catchment received extra input of 10% by "cloudstripping" (Weaver 1972) with chemical composition as determined by Trinidad Pizarro in [33]; [y]phosphate-phosphorus; [z]nitrate-nitrogen, input as determined at top end of watershed; input of ammoniacal nitrogen of similar magnitude ([33]); if concentrations of total nitrogen in rainfall as reported for El Verde at 500 m (1970) are extrapolated to entire watershed an input of ca. 23 kg ha^{-1} yr^{-1} is obtained.

[34]Aitken et al. (1972); [35]Edwards (1982)
Lower Montane Rainforest at 2450 m; andesitic volcanic tuffs; inceptisols; amount of streamflow estimated by subtracting forest ET (as determined from regional ET-elevation-rainfall relationship[34]) from rainfall; composition of streamflow based on single baseflow sample taken in July 1971; baseflow composition at that time of year probably within 10% of annual mean (Bruijnzeel 1983a); nutrient input based on bi-weekly readings of rainfall amounts in two clearings and 9 samples each between December 1970 and August 1971; 1 cm wire mesh in funnels, no preservatives.

Table 3 continued

[36]Jordan (1969); [37]Edmisten (1970); [32]Brown et al. (1983)
Lower Montane Rainforest at 500 m; andesitic rocks; acid topsoil overlying yellow clay and red/yellow clay/rotten rock; rainfall (2 collectors above canopy) and streamflow (catchment area unknown) sampled weekly between October 1967 and September 1968; [A]as given in [32], Jordan (1969) may have used different values (not specified); [B]based on limited data collected in December-February by [37].

[38]King & Yang (1984); [39]Cheng et al. (1987)
Warm-temperate coniferous forest on slopes and warm-temperate rainforest in valleys; sites at 2550 m and at 24° N.L.; slates; brown podzolic soils with silty to sandy loam texture; catchment areas 144 ha (PL-11) and 238 ha (PL-12); vegetation in PL-11 denser; rainfall and streamflow monitored since 1969; sampling for chemical analysis between January 1981 and December 1982 (anions 1982 only) with precipitation ca. 8% above longterm means; [C]inorganic forms of the element only; total outputs of soluble nitrogen ca. 2.1 kg ha^{-1} yr^{-1} for both catchments.

[40]Naprakob et al. (1976); [41]Watnaprateep (1984)
Evergreen hill forest dominated by Castanopsis; granodiorite?; "latosol" of sandy clay loam texture, probably very poor; catchment area 65 ha; rainfall and runoff monitored since 1965; weekly sampling of rainfall and streamflow in 1974-1975; no further experimental details given; concentrations in rainfall rather high, the ones quoted for 1969 are lower and similar to those of Manokaran (1980).

[42]Steinhardt (1979)
Lower Montane Rainforest ("cloud" forest); silty and clayey sedimentary rocks; humitropepts with variable drainage; drainage rate approximated by subtracting forest ET (as determined by energy balance method) from rainfall; chemical composition of drainage water determined with three suction lysimeter plates with suction set at levels in surrounding soil (twice/week); [D]nutrient outputs considered underestimate by investigator himself; rainfall sampled from plastic gauges at least weekly between December 1983 and November 1984 (rainfall 10% higher than normal); [E]samples considered contaminated with insect debris by investigator.

from various publications. Although perhaps a rather bold thing to do, the exercise was considered justified in view of the limited amount of published information available for the montane tropical environment and the rather flimsy hydrological foundations on which some often quoted budgets (e.g. no's 1, 3, 7, 16 and 19 in Table 3) are based. Details on measuring and computation procedures for all studies quoted in Table 3 are given in footnotes.

Sites were subdivided as follows: first, a distinction was made between lowland forests (n=19) and montane forests (n=6). The lowland sites were then grouped according to the nature of the substrate, i.e. soils with moderate to very low fertility (n=12), and soils with moderate to high fertility (n=7). Most of the forests belonging to the low fertility group are on Ultisols and Oxisols, with one site on Spodosols.

The variation in reported nutrient fluxes, not only per element and fertility group but also per lithology and even for particular study sites (sic!), is considerable (Table 3).

For example, reported calcium and magnesium inputs for the Oxi/Ultisol group range from barely detectable at sites no. 10 and 11 in the Central Amazon to 42 and 17 kg ha^{-1} yr^{-1} respectively at site no. 12 (Peninsular Malaysia). Similarly, losses for these elements vary between negligible in the Amazon and 47 and 30 kg^{-1} ha^{-1} yr^{-1} respectively at site 6 (Ivory Coast). Although such differences might theoretically reflect differences in rainfall amounts and regimes, and/or bedrock type and degree of weathering in contrasting environments, this becomes less probable when finding strikingly different net losses or gains for catchments experiencing comparable climatic and geological conditions (e.g. sites 4, 7 and 12, which are all on deeply weathered granites in the Far East, or sites 3 and 10/11, all in the Amazon and on very infertile substrates).

Rather than taking such findings at face value, one would be well advised to investigate the possibilities of contamination/enrichment of precipitation samples by regional dust (Roose 1981; Crowther 1987a; Yusop et al. 1989), fire (Edwards 1982; Jordan 1982) or organic debris (Steinhardt 1979). Also, difficulties with the analysis of highly dilute water samples (Turvey 1974; Jordan 1982; Poels 1987) and the underestimation of streamflow amounts due to basin leakage (all Amazonian sites with sandy valley fills (Bruijnzeel 1989b; cf. Abdul Rahim & Kasran 1986) or the use of too high an estimate for forest ET (producing an underestimate of the mount of drainage; Jordan 1982; Parker 1985) sometimes influence the results.

It is perhaps significant in this respect that Bruijnzeel (1989a) concluded from a comparison of average elemental concentrations of bulk precipitation at a number of stations in the humid tropics, that the results often seemed to reflect the rigidity of sampling procedures (including positioning of collectors) rather than environmental factors like proximity to an ocean, volcanism or climatic seasonality (see also Lewis et al. 1987). It is quite possible, therefore, that many studies of elemental fluxes in tropical forest areas have overestimated atmospheric nutrient inputs, although the effect will be moderated to an unknown extent by the fact that contributions of dry deposition on forest canopies are largely unaccounted for by traditional collecting devices. Further work on this aspect is as desirable as difficult (White & Turner 1970). Some progress however has been made in this respect in relation to such problems as acid deposition and forest dieback in the temperate zone (e.g. Prupacher et al. 1983; Gosz et al. 1983; Lovett & Lindberg 1984; Lindberg et al. 1986; Delleur 1989; cf. Lewis et al. 1987).

Detecting procedural deficiencies is one thing, correcting them quite another. However, in some cases better hydrological information and alternative rainfall chemistry data for the same or a nearby location was used to obtain more realistic results. For instance, the use of an alternative data set for site 3 (San Carlos, Venezuela) transformed this apparently rather vigorously nutrient accumulating forest (Jordan 1982) into a system with marginal losses, which is much more in line with results obtained for similar forest on very poor soils elsewhere in the Amazon (Brinkmann 1983; Franken & Leopoldo 1984; sites 10 and 11; see footnotes in Table 3).

Repeating the procedure for another apparently nutrient accumulating system (site 12, Bukit Berembun, Malaysia) yielded a similar improvement in that the corrected net solute outputs were much closer to those reported for granitic terrain in Hong Kong (site 4; Lam 1978) than the previous ones (Table 3).

As such, it remains to be seen whether the Surinam forest (site 9 in Table 1; Poels 1987) is indeed accumulating calcium and potassium to the extent claimed by the investigator, especially when taking into account the difficulties that were met in collecting and analyzing representative precipitation samples as well as the possibility of unrecorded deep flow through the sandy valley fills (cf. Table 1).

3.4.2 Lysimetry- versus catchment based estimates

The discrepancies between estimates of hydrologic nutrient outputs based on soil- or groundwater sampling on the one hand and streamflow sampling on the other, become readily clear from the results obtained at several locations where both techniques were used (sites 1, 6, 8, 15 and 17, to a lesser extent also site 14).

From the data reported for the Spodosol site at San Carlos (site 1; Herrera 1979), one could either conclude that the forest is losing nutrients (zero-tension lysimetry) or is accumulating them (samples taken from a small stream). An alternative computation by the present writer also suggested slight accumulation (see footnotes in Table 3). Conversely, an intensive study of precipitation and streamflow chemistry for the much larger (47500 km^2) Caura river basin, a "blackwater" area some 500 km northeast of San Carlos draining equally infertile soils, indicated high gross and net losses of calcium, magnesium and potassium. Phosphorus was retained (site 2; Lewis 1986; Lewis et al. 1987).

Similarly, large losses of calcium, magnesium and potassium have been reported for an old secondary forest in the Ivory Coast (site 6) on the basis of zero-tension lysimetry by Roose (1981), whereas the same forest would be accumulating these elements on the basis of (headwater) stream sampling. The opposite pattern was reported for phosphorus. Roose's findings were confirmed for calcium, but not for magnesium and potassium, by Bernhard-Reversat (1975) for a similar forest nearby (site 8).

How are such apparently conflicting results to be explained? In the case of nutrient poor systems, where the surface rootmat has been shown to be an efficient filter for incoming nutrients (Stark & Jordan 1978; Brinkmann 1983), one may indeed expect a slight decrease in soil water solute levels with depth as further nutrients are taken up by roots in the subsoil without being replaced by actively weathering minerals. As such, somewhat larger nutrient losses may be obtained on the basis of (zero-)

tension lysimeters installed at relatively shallow depths in these depleted soils than on the basis of streamwater chemistry (cf. the results obtained by Russell (1983) and Poels (1987), sites 5 and 9 respectively), provided the stream has not reached the *active weathering front* (i.e. the contact between weathering mantle and fresh bedrock (Bruijnzeel 1989b).

As shown by the Caura basin study (cf. site 20 in Table 3), streams draining areas with infertile soils may still carry considerable amounts of nutrients *once the stream has exposed the fresh bedrock*. The point is neatly illustrated by the chemical analyses for a deep Oxisol on granite in Peninsular Malaysia presented by Burnham (1989). Soil cationic concentrations were very low throughout the profile (which included a substantial portion of weathered rock) and rose sharply in a narrow band close to the fresh rock at a depth of ca. 10 m. Since streams draining this type of terrain carry fair amounts of cations (Douglas 1967b; Kenworthy 1971; Abdul Rahim & Yusop 1986), it cannot be excluded that the latter derive to a large extent from this narrow zone of active weathering (Bruijnzeel 1989b).

Nutrients released by rock weathering at such depths are essentially out of reach of the root network (Baillie 1989). Therefore, although clearly a loss to the ecosystem at large, these nutrients do not play a role in the nutrient economy of the forest itself. It follows that in such cases the dissolved nutrient load of a stream is not representative of the nutrient loss experienced by the vegetation. Experimental work is needed to test the above hypothesis and may usefully combine observations of root distribution (Baillie & Mamit 1983) with determinations of soil- and soil water composition throughout the weathering profile at different topographic positions (Verstraten 1980; Bruijnzeel 1983a).

Discrepancies in nutrient exports obtained via lysimetry or streamflow sampling should be relatively small in the case of deep and more fertile soils, where contrasts in nutrient contents of bedrock and soil material are less pronounced and competition for nutrients in the topsoil is less severe. Nevertheless, water emerging as springs in such areas is usually somewhat more concentrated than soil water sampled higher up in the profile (Bruijnzeel 1983a; Parker 1985), reflecting differences in residence times. Interestingly, ecosystems on substrates in which only a certain nutrient is particularly scarce, e.g. potassium in limestones or ultrabasic rocks, exhibit an oligotrophic cycling strategy (Jordan & Herrera 1981) with respect to that nutrient, but not for other elements. For example, Crowther (1987b) found much higher potassium concentrations in soil water intercepted by zero-tension lysimeters at the (rather shallow) soil-bedrock (limestone) contact in West Malaysia than in deep groundwater seeping into the underlying caves (sites 15 and 17 in Table 3). Apparently, deep roots in fissures and joints were able to extract considerable amounts of potassium from the percolate. Discrepancies between the two types of samples were generally much less for calcium and magnesium, which were abundantly available. Similarly, potassium concentrations in the soil solution (as sampled by suction lysimetry) decreased rapidly with depth in montane rain forests on ultrabasic rocks in Sabah, with very low amounts present in stream water (L.A. Bruijnzeel & M.J. Waterloo, unpublished data). Although seepage water was in contact with the fresh rock in both the above examples, no rise in potassium concentrations was recorded, reflecting the inability of the rock to supply this element in significant quantities.

The examples given sofar do not only confirm the methodological limitations referred to in the previous section, they also provide some guidance as to which method (the lysimetric or the catchment approach) is the more representative when it comes to evaluating nutrient losses from forest ecosystems. It would seem as though carefully selected small, yet watertight, catchments, supplemented by hillslope plots if the spatial variation of soil and vegetation requires so, and monitored for a number of years to account for climatic variability, could give the best estimates of ecosystem nutrient losses (Clayton 1979). Estimates based on various forms of lysimetry alone should be treated with caution in view of the above-mentioned problems related to the estimation of drainage and average soil water composition.

Highly infertile soils present a special case in that different approaches are needed, depending whether one is primarily interested in the nutrient loss from the biologically active portion of the ecosystem or from the system at large. In the former case, either the lysimetric or the small catchment approach may do, in the latter a sufficiently large catchment area is needed.

3.4.3 Patterns of calcium, magnesium and potassium losses

During the discussion presented thus far one may well have wondered to what extent it is possible to derive any meaningful generalizations from the varied data in Table 3. However, scatter plots of annual calcium, magnesium, and potassium outputs against corresponding amounts of streamflow for most (small) catchment studies listed in the Table show that this is indeed feasible to a certain extent (Figures 12a-c). Despite some overlap one can clearly distinguish four groups with contrasting nutrient export patterns. Figure 12d presents the range of values associated with the respective groups.

The sites of the *Spodosol/highly depleted Oxisol group* (mainly Amazonian studies, also site 24) consistently exhibited very low losses of calcium and magnesium, and often of potassium as well. The highest values in this group were reported for "extremely poor" (Buschbacher 1984) Ultisols near San Carlos de Rio Negro (alternative computation, Table 3). Since these estimates are based on zero-tension lysimetry (Buschbacher, unpublished data, quoted by Medina & Cuevas 1989), they may represent slight overestimates, although it may well be argued that they reflect a somewhat less weathered substrate, the soil after all having been classified as an Ultisol rather than an Oxisol. As such, this site may represent a transition to the second group (see below).

Nutrient losses for the *Oxisol/Ultisol group* (n=5) are somewhat larger and more variable than those observed for the most infertile sites, although there is some overlap for potassium (Figure 12d). Nevertheless, the present group average for potassium (about 11 kg ha^{-1} yr^{-1}) is much higher than that for the first group (< 2.7 kg ha^{-1} yr^{-1}).

Still larger losses of calcium and magnesium have been recorded for sites that are largely on *Inceptisols* (n=5). Although again there is an overlap with the preceding group for potassium, average values for the two groups are quite different at 23 versus 11 kg ha^{-1} yr^{-1} respectively.

Finally, very high losses of calcium have been reported for forests on calcium-bearing rocks with *Mollisols/Vertisols* (sites 15, 17, and 19 in Table 3). The corresponding outputs of magnesium (variable) and potassium (low and similar to the Oxisol/Ultisol group) reflect the nature of the underlying rocks (cf. sites 16 and 23, which are both on magnesium bearing rocks and hence exhibit fairly high losses for this element).

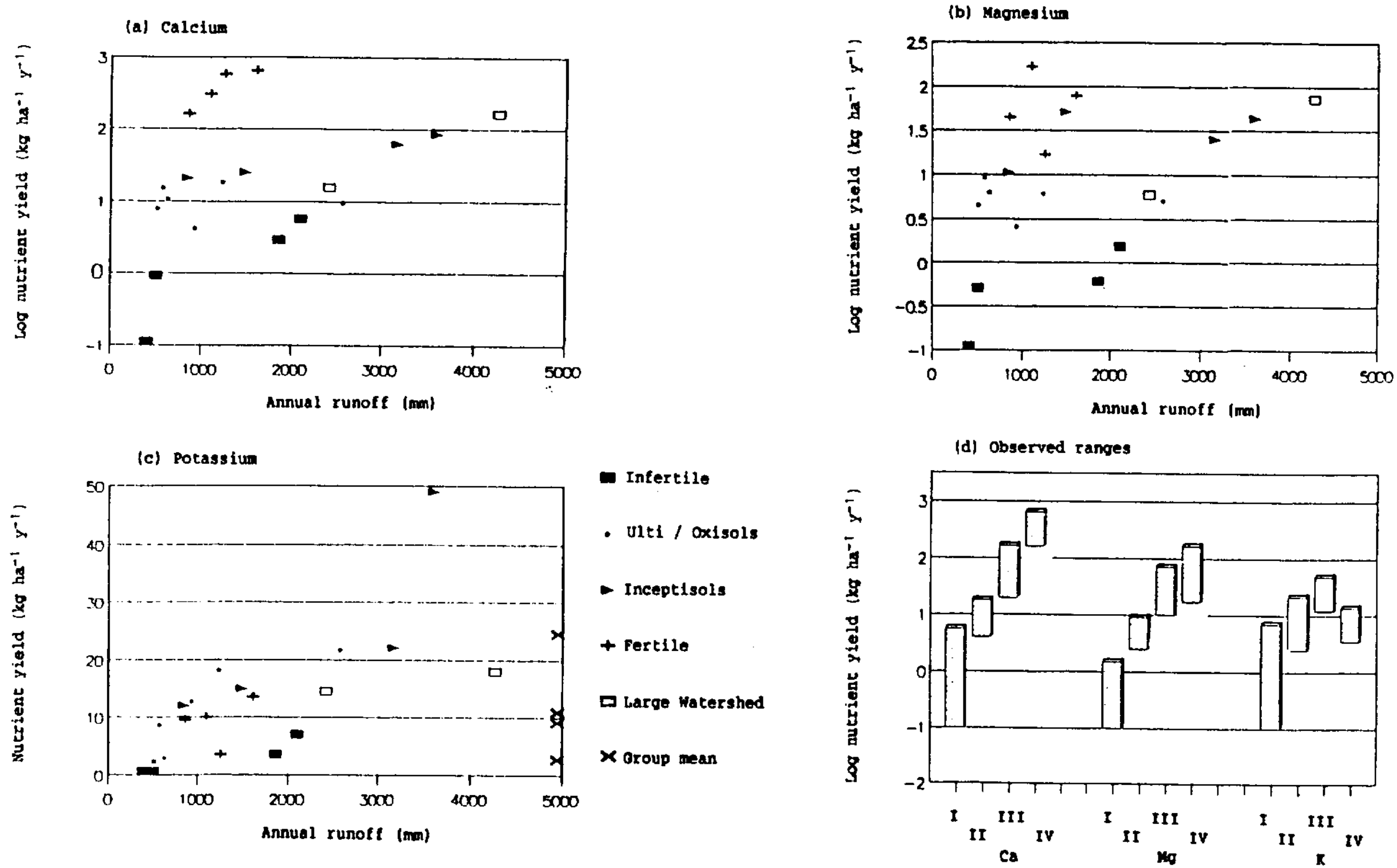

Figure 12. Scatter plots of annual runoff versus nutrient yield for 20 selected (sub)tropical forest ecosystems (a) calcium, (b) magnesium, (c) potassium, (d) observed ranges.

Patterns of calcium, magnesium or potassium losses from *Montane forests* do not seem to deviate from those for lowland forests (Table 3 and Figure 12; see also the next section).

3.4.4 Phosphorus and nitrogen

Phosphorus appears to be accumulating in almost all studies quoted in Table 3, reflecting the very low mobility of the element. It has been suggested that most of the phosphorus released by weathering becomes tied up in organic and ferro-aluminium compounds in the soil (Sanchez 1976; Clayton 1979). As such, one cannot expect the high amounts of phosphorus reportedly released by the weathering of the volcanic tuffs at site no. 13 (and largely incorporated in the rapidly growing vegetation; Bruijnzeel 1983a,b) to show up in streamflow after removal of the vegetation (cf. Chapter 5). Incidently, this low mobility of phosphorus, apart from the various possibilities mentioned earlier, also renders the extremely high phosphorus exports reported for San Carlos (no's 1 and 3 in Table 3) highly unlikely.

Nitrogen also represents a special case in that its biogeochemical cycle includes a number of processes that involve the element in its gaseous form (Odum 1971). A significant portion of nitrogen inputs to the forest ecosystem occurs via biological fixation. The representativity of the data on (atmospheric) inputs and (hydrologic) outputs of nitrogen presented in Table 3 is limited still further by the variation in nitrogen components covered by the various studies as indicated in the respective footnotes. Conclusions of net losses or gains based on inorganic nitrogen constituents alone are easily overturned by the inclusion of the organic fraction (Brinkmann 1983). Therefore, the remark of Vitousek & Sanford (1986) with respect to the possibility of detecting any patterns for tropical forests on the basis of available data is especially valid for this element and no further attempt at analysis has been made here.

More or less comprehensive estimates of the various inputs and losses of nitrogen have been made for only two tropical forests, the Oxisol site at San Carlos (site 3 in Table 3; Jordan et al. 1982) and the lower montane rain forest at El Verde, Puerto Rico (site 22; Edmisten 1970). Although limited in number, the results neatly illustrate the limitations of the small watershed technique in this respect.

For example, only 42 per cent of the total nitrogen input at San Carlos entered the forest via bulk precipitation, whilst 58 per cent was supplied by biological fixation. Also, hydrologic losses made up 83 per cent of the total loss, with denitrification accounting for the remaining 17 per cent (Jordan et al. 1982). In other words, the (inorganic) nitrogen budget at this site was transformed from a net loss of 2.6 kg ha^{-1} yr^{-1} (Table 3; hydrologic inputs and outputs only) to a net gain of 8.6 kg ha^{-1} yr^{-1} when all processes were taken into account.

It is difficult to envisage, however, how a forest which is presumably in a state of dynamic equilibrium (Uhl 1982), would be accumulating this amount of nitrogen. Alternately, this result may reflect the fact that concentrations of organic nitrogen were not determined. Elsewhere in the Amazon ("clear water" areas near Manaus), organic nitrogen constituted 78 and 88 per cent of the total hydrologic gains and losses of nitrogen respectively (Brinkmann 1983).

In contrast to the observations at San Carlos, those at El Verde were made during a rather short period with relatively low rainfall totals (Edmisten 1970). Fixation on leaf surfaces and by root nodules accounted for 86 per cent of the total nitrogen input, i.e. only 14 per cent was supplied via precipitation. Also, losses in drainage water (29 kg ha^{-1} yr^{-1}) were much smaller than those associated with denitrification (ca. 56 kg ha^{-1} yr^{-1} or 66 per cent of the total loss). Again, an apparent net loss of 15 kg ha^{-1} yr^{-1} (hydrological budget; Table 3) was turned into a net gain of 17 kg ha^{-1} yr^{-1} after including biological transfers of nitrogen.

Montane rain forests, like "heath forests", are generally considered to be short of nitrogen, although this contention is based on analyses of individual processes, such as litterfall and mineralization rates in the soil (Grubb 1977; Vitousek & Sanford 1986; Marrs et al. 1988) rather than on a quantification of all processes involved (Edmisten 1970). In view of the apparently conflicting evidence, further work, preferably of an integrated nature, is desirable.

Summarising, nutrient losses from tropical forest terrain are primarily governed by the nature of the geological substrate and the degree to which this has been weathered. A distinction has to be made between losses from the ecosystem at large and losses from the biologically active portion of the ecosystem, especially for oligotrophic sites.

The present chapter concludes our description of the undisturbed tropical forest environment. The next two chapters will discuss the hydrological and chemical responses of the ecosystem to various kinds of disturbance by man.

4 HYDROLOGICAL IMPACTS OF (DE)FORESTATION

4.1 Introduction

Hunter-gatherer societies have lived in tropical rain forest regions for thousands of years without destroying the forest (Lea 1975). Although some parts of the forest were cleared long ago for permanent agriculture (e.g. the alluvial plains of Asia, which formerly carried swamp forest, for the cultivation of irrigated rice), for a long time the most extensive farming in rain forests all over the tropics has been by "shifting cultivation", also known as "slash and burn agriculture" (Whitmore 1990).

Shifting cultivation has been identified as the single most important reason for loss of tropical forest before the 1980's (Lanly 1982). It was estimated to be responsible for about 35 percent of all deforestation in Latin America, whilst the corresponding figures for Africa and Asia read 70 and 50 percent respectively (Hadley & Lanly 1983). Jackson (1983) estimated that about 150 million people are involved in traditional agriculture in the tropical forest zone.

Basically, shifting cultivation consists of (manually) felling a patch of forest, usually at the start of the period of least rainfall, allowing the material to dry, burning the slash shortly before the rainy season, and planting rapidly maturing crops in the ash (Plate 3). After one or two crops have been harvested, yields diminish and the original field is abandoned to forest regrowth (the so-called "bush fallow"). The farmers then move on to another piece of forest and the cycle is repeated. Since it is easier to fell and burn secondary forest than virgin jungle, repeated rotation through an area is often preferred by shifting cultivators to continual movement into new areas (Watters 1971; Scott 1987). A number of variations on this basic theme can be distinguished (Chuasuwan 1985).

In contrast to the view prevailing a few decades ago (FAO 1957), it is now more or less generally accepted that shifting cultivation is a form of agriculture that is sustainable under the general climatic and edaphic conditions of the humid tropics, as long as it is practiced within the limits of the ecosystem's capacity to regenerate. However, when either the cropping period is extended too long (impairing forest succession: Uhl 1987), or the bush fallow period becomes too short (limiting the build-up of nutrients in the vegetation to be released upon burning), the system will degrade (Zinke et al. 1970; Sanchez 1976; Hatch 1983; Scott 1987; see also Chapter 5).

Although rain forests are capable of supplying a host of minor products (Lea 1975; Myers 1988a) and have done so for a long time, there has been a rapid change in the way these forests are valuated by modern man. Nowadays, rain forests are mainly regarded as a source of timber. For example, just before the second world war, the relative importance of timber and minor forest products traded from Indonesia was 55 to 45 per cent. Today, it is 95 to 5 per cent (Whitmore 1990).

Estimates of the rate at which moist tropical forests were being altered by man in the early eighties lie around 12 million ha yr^{-1}, 63 per cent of which (7.5 million ha) was cleared, with the remainder (4.4 million ha) "selectively" harvested. Another 4 million ha yr^{-1} were reportedly cleared in the more seasonal tropics (Lanly 1982). These figures are probably conservative to the extent that national reports to FAO may well have been over optimistic. In addition, the great forest fires occurring in Borneo in August - October 1982 and in March - May 1983 (damaging more than 4 million ha of (partially logged-over) forest; Malingreau et al. 1985) and the deliberate burning of 16 million ha (!) of forest along the southern fringe of Amazonia in 1987 and 1988 for the creation of pastures (Whitmore 1990) are not included in the above overall estimates.

The causes and patterns of forest clearance vary considerably between regions and a detailed analysis of these is beyond the scope of the present report (see overviews by Myers 1980; Lanly 1982; Whitmore 1990). Now that satellite imagery is becoming more widely available, chances for adequate monitoring of rates of forest removal are improving (Myers 1988b). Recent reports on local or regional situations include those by Gentry & Vasquez (1988) (Peruvian Amazon), Fearnside (1987) (Brazilian Amazon), Malingreau & Tucker (1988) (southeastern Brazil), Eyre (1987) (Jamaica), Sader & Joyce (1988) (Costa Rica), Pullan (1988) (West Africa), Hirsch (1987) (Thailand), Quinnell & Balmford (1988) (Philippines) and Smiet (1989) (Indonesia).

As indicated earlier, the general term "deforestation" is rather meaningless as a descriptor of land-use change and each case needs to be defined properly (Bruijnzeel 1986; Hamilton 1987). In this and the next chapter, three levels of intensity of forest disturbance, viz. low, intermediate and high (Jordan 1985), will be distinguished when discussing the environmental impacts of "deforestation".

Low-intensity types of disturbance (mainly dealt with in Chapter 5) include such small-scale and short-lived events as natural tree falls and small clearings. Apart from the already mentioned forest fires and slash and burn agriculture, both of which generally produce a temporary effect, selective logging of forests may also be ranked as a disturbance of (at least) intermediate intensity, depending on the volume of timber removed and the type of equipment used (Horne & Gwalter 1982; Plates 4-8).

The partial removal of timber from a forest stand may be interpreted as the creation of a large number of variably sized gaps (Jordan et al. 1985). Since modern logging techniques are generally highly mechanised, such an operation implies the creation of access roads, skid (or "snig") tracks and landings, and in doing so a considerable portion of the area that is to be logged will become disturbed (Burgess 1971, 1975). Often, the lay-out, construction and maintenance of logging roads, etc. is poor and former skid tracks and landings may remain compacted for many years after their creation (Malmer & Grip 1990; Van der Plas 1990) and will influence runoff and regeneration patterns accordingly (Plates 6 and 7).

As such, selective logging will usually have to be classified as a disturbance of moderate intensity at least, also because felling and extraction of large trees may produce so much damage to the surrounding vegetation that regrowth may be too slow for further profitable exploitation (Burgess 1971; De Graaf 1986). Finally, roads make logged-over forests more accessible to settlers, hunters, etc., thereby increasing the risk of further degradation (Wyatt-Smith 1987).

Generally, a forest subjected to one of the above-mentioned types of disturbance may recover to its previous state if left alone for a sufficiently long period (Saldarriaga 1987; Riswan & Kartawinata 1988). Clearly, this is not the case when forest is converted to permanent agriculture (grazing, cropping, extractive tree crops) or production forestry and these must all be classified as disturbances of high intensity.

Forest land can be cleared in a number of different ways, each characterised by a certain degree of soil disturbance. Couper et al. (1981)

compared the effects of several clearing methods, ranging from traditional slash and burn through modernised manual clearing (using chainsaws) to highly mechanised techniques, in terms of man hours and energy expenditure, whereas Lal (1981) reported on surface erosion rates associated with the various techniques as observed at the same site. Although manual clearing was slowest and the most expensive of the methods investigated, soil erosion in the first year after clearance amounted to only 0.4 t ha^{-1} (Lal 1981). Conversely, clearance by means of crawler tractor with a shear blade attached, although the most rapid and economical, induced an erosion rate of almost 4 t ha^{-1} yr^{-1}. Finally, crawler tractors with tree pusher/root rake attachments were more expensive to use than the ones equipped with a shear blade only and produced an erosion rate of more than 15 t ha^{-1} yr^{-1}. All values cited were for no-tillage agriculture during the first year after clearing (Couper et al. 1981; Lal 1981).

Similarly, Van der Weert (1974) and Seubert et al. (1977) drew attention to the negative effects of mechanised clearing on root development and therefore on agricultural production, mainly through increased soil compaction. More recently, the experiments of Dias & Nortcliff (1985a) revealed a close correlation between the number of tractor passes over an oxisol in Amazonia and the resulting degree of soil compaction. Uhl et al. (1982, 1988b) reported slow regeneration of natural regeneration following clearing by bulldozer in areas underlain by Podzols and Oxisols in Latin America.

Although there can be no doubt that manual methods of forest clearing are far superior to most modern methods of clearing in terms of damage done to the forest floor (Dias & Nortcliff 1985b), there will often be no choice but to resort to mechanical means. However, as pointed out by Couper et al. (1981), bringing new land under cultivation should not just be done with the objective of carrying out as much as possible as "cheaply" as possible, since this approach could well result in irreversible damage to the land. The short-term gains would then be more than offset by the costs needed to restore or maintain soil productivity in the long term (Van der Weert 1974; Seubert et al. 1977; Ollagnier et al. 1978).

Martin (1970) and Van der Weert (1974) have given a number of simple suggestions as to how soil damage during clearing could be minimised. These include the proper choice of the timing of the operation (preferably during times of low soil moisture levels as wet soil is easier to compact), the avoiding of the use of tree-pushing/root raking equipment wherever possible (i.e. leave stumps to rot), and the minimising of the number of tractor passes during windrowing by optimising the distance between windrows and burning the slash before windrowing. By planting crops close to the windrows the reduction in windrow spacing should not affect accessibility in the newly established plantation too seriously.

Couper et al. (1981) stressed the importance of employing skilled tractor operators since bulldozers were originally designed to move soil rather than clear forest. In addition, the regular sharpening of tractor shear blades greatly increased the efficiency of the method. Provided all of the above cautionary measures are taken, Couper et al. (1981) reckoned shear blade clearing to be capable of providing a suitable base for sustained yields in the future.

Although sky-line logging (also called "high-lead yarding") has been shown to be capable of minimum disturbance of the surface in steep terrain (Pearce & Griffith 1980; Plate 8), the degree of damage done to the vegetation surrounding the harvested trees makes this technique more suitable for use in clearcutting than in selective logging operations.

In the following, the effects of partial and complete forest removal c.q. planting on climate (notably rainfall; section 4.2), water yield (section 4.3) and its seasonal distribution (section 4.4) as well as on sediment production (section 4.5) will be looked at in detail.

4.2 Effect on rainfall

Differences in micro-climatic conditions near the ground in tropical rain forests and large clearings are well-documented (see Richards (1952) and Schulz (1960) for excellent discussions of the early literature; also Pinker 1980; Lawson et al. 1981; Ghuman & Lal 1987).

In general, near-surface conditions in clearings are much more harsh than inside the forest, with greater insolation, higher maximum temperatures and vapour pressure deficits and, especially, much higher soil temperatures, resulting in a strongly increased evaporative demand of the atmosphere (Figure 13). The direct ecological (e.g. decomposition of organic matter, microbial and soil faunal activity, etc.) and site-hydro-

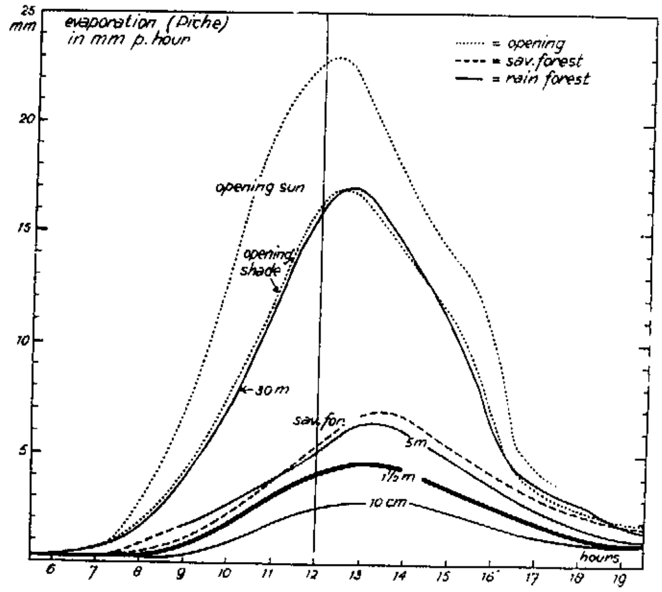

Figure 13. Daily march of Piche evaporation at several heights in rain forest, savana forest and a large clearing during the dry season of 1957 in Surinam. All instruments shaded against direct sunlight except one (after Schulz 1960).

logical (e.g. topsoil infiltration characteristics and erodibility) consequences have been discussed at length by Lal (1987).

However, an evaluation of any climatic changes brought about by forest removal requires a comparison of micro-climatic conditions over the newly cleared land with those prevailing at the top of the forest canopy rather than inside the forest (Thompson & Pinker 1975; Shuttleworth et al. 1985). As shown by Schulz (1960) and Thompson & Pinker (1975), temperatures and vapour pressure deficits experienced above a rain forest canopy are very similar to those recorded at screen height in large clearings (cf. Figure 13), although large differences may be found closer to the ground surface (Thompson & Pinker 1975; Pinker 1980).

There exists an important difference, however, in radiative characteristics exhibited by a rain forest canopy and various types of other surfaces. Tropical rain forests typically reflect about 12 per cent of the incoming short-wave radiation (Oguntoyinbo 1970; Pinker et al. 1980; Shuttleworth et al. 1984) whilst corresponding amounts for grassy clearings are close to 15 per cent (Pinker et al. 1980) or even 20 percent in the case of various agricultural crops (including well-watered grassland; Montény 1986). Therefore, a different partitioning of available energy between warming up of the boundary layer (i.e. sensible heat) and evaporation (i.e. latent heat) is to be expected upon conversion of tropical forest to grassland or agricultural crops, also because of differences in rooting depth and hence capacity to exploit soil moisture during dry spells (Shuttleworth 1988a). This in turn, if effected over a sufficiently large area, may affect local and regional circulation patterns of air and therefore rainfall (Salati & Vose 1984; Montény 1986; Dickinson & Henderson-Sellers 1988; Shukla et al. 1990; see below).

If, on the other hand, the forest is replaced by a vegetation type with rather similar radiative and evaporative characteristics, such as rubber or oil palm plantations, presumably the effect would be much less (Montény 1986). As such, it cannot be stressed enough that it is imperative to state the nature of "deforestation" when discussing these matters (Hamilton 1987).

The issue of the influence exerted by forest vegetation on the amount of rainfall it receives has already been touched upon in section 2.3.1. Two different approaches have been followed by investigators trying to resolve the question, which could be termed the "direct" and the "indirect" approach. The former analyses time series of rainfall and vegetation data whilst the latter uses computer simulation. Results obtained with both methods are reviewed below.

Circumstantial evidence for decreased rainfall at individual or a group of measuring stations in the tropics abounds in the literature and is often ascribed to concurrent "deforestation" (see numerous examples in Meher-Homji 1988). Unfortunately, many of these analyses cannot be considered very rigorous in that either the number of gauges or the period of observation taken into account were limited. In addition, synoptic considerations are rarely included in the analysis (cf. Dickinson 1980).

Mooley & Parthasarathy (1983) examined the occurrence of above- and below-average annual rainfalls between 1871 and 1980 for 306 stations all over the Indian subcontinent. They were unable to detect any trends or oscillations that were statistically significant and concluded that during the period under consideration annual rainfall totals over India were distributed randomly in time. Although it could be argued that most of the stations used in this analysis had lost their forest cover a long time ago (Meher-Homji 1988), the occurrence of a very wet or dry year seemed to be related to the degree to which depressions were able to penetrate the subcontinent in a westward direction. Wet years showed a distinctly higher proportion of depressions moving west of 80° E.L. (Mooley & Parthasarathy 1983).

In addition, the regional distribution of rainfall over India appeared to be strongly related to the location of the "monsoon trough", a zone of relatively low pressure which normally runs between South Bengal and Rajasthan (Ramaswamy 1962). The trough may shift towards the foothills of the Himalaya, producing a marked decrease in rainfall south of it (i.e. over the subcontinent) and a distinct increase in precipitation over the Himalaya (Dhar et al. 1982). These so-called "breaks" in the monsoon were shown to occur about three times more frequently during "drought" years than during "flood" years, whilst the average length of "break" was two to three times higher during dry years as well (Mooley & Parthasaraty 1983).

The above phenomenon could well explain a number of observations of locally decreased rainfall quoted by Meher-Homji (1988) but not such long-term persistent trends as that described for upland Sri Lanka by Madduma Bandara & Kuruppuarachchi (1988) (ca. 500 mm yr^{-1} between 1878 and 1970 in an area where a substantial portion of forest has been converted to tea plantations over the years). Incidently, Werner (1988), discussing the dieback of montane forest in the same area after a severe drought in 1976, reported that much higher rainfall totals have occurred again from 1984 onwards and that the forest was on the road to recovery. Since ET of mature tea plantations is not dramatically lower than that for montane forest (Blackie 1979b; Table 4), the above decrease in rainfall can hardly be attributed to the change in land use alone (Madduma Bandara & Kuruppuarachchi 1988). Rather, one must think of changes in the movements of the equatorial trough (cf. Arulanantham 1982).

Clearly, there is a need for rigorous statistical analysis of long-term rainfall records for carefully selected representative stations in relation to concurrent data on vegetation cover. Ideally such work should take into account synoptic situations as well. Arguably, data from areas with relatively high rates of sub-recent "deforestation" such as Ivory

Coast, Costa Rica, Thailand or Sumatra (Jackson 1983) could be used in such an analysis.

Fleming (1986) analysed time trends of annual rainfall totals for ten stations in Costa Rica with records ranging between 28 and 95 years. Low-land sites all exhibited a decrease in precipitation with time whereas virtually all stations in the hills showed an increase, although the statistical significance of the trends was weak. However, the slope of the regression line relating annual precipitation with time increased significantly with elevation. Fleming (1986) hypothesised that the large-scale conversion of semi-deciduous forest in the western lowlands of the country to pastures and dry-land cropping had brought about changes in air mass circulation that favoured the discharging of precipitation over the hills (cf. Meher-Homji 1980; Nooteboom 1987. It would be interesting to examine whether any long-term trends in spatial variations in sunshine duration exist in the area as well to see whether patterns of cloud cover (e.g. using satellite imagery) have changed accordingly.

Tangtham & Sutthipibul (1989) compared regionally averaged data on rainfall amounts and occurrence for thirty six stations in northeast Thailand with changes in forest cover over the period 1951-1984. On a year to year basis there was no correlation whatsoever between any rain-fall parameter and the percentage of remaining forest cover, although an-nual rainfall totals generally exhibited a weak negative trend during the period under consideration. However, when annual rainfalls were expressed as 10-year moving averages there was a significant negative correlation with remaining forest area whilst a positive correlation was found bet-ween the latter and the number of rainy days. In other words, showers tended to become more frequent and smaller, although Tangtham & Sutthi-pibul (1988) were quick to point out that the "effect" of deforestation was still within one standard deviation of the means of the respective time series.

An opposite trend, i.e. less showers but of higher intensity, has been suggested by long-term observations of daily rainfall on private rubber estates in Peninsular Malaysia (UNESCO 1978). More work is needed.

The "indirect" approach (computer simulation of climatic effects of land-use changes) avoids the problem of high spatial and temporal variability in tropical rainfall to some extent but has its own share of problems.

Henderson-Sellers (1987) considered a realistic estimate of climatic effects of tropical deforestation "nearly impossible" for two main reasons: (1) the lack of reliable data on the nature and extent of "deforestation" and (2) limitations of the simulation models themselves and inadequacies in the statistical methodology for interpreting results obtained through them. She compared the results of four early simulations for the Amazon basin (Potter et al. 1975; Lettau et al. 1979; Henderson-Sellers & Gornitz 1984; Wilson (1984) in Henderson-Sellers 1987) and showed these to be rather contradictory.

For example, changes in temperature ranged from a decrease of about 0.5 °K through no change at all to an increase of about 0.5 °K, whereas changes in annual precipitation ranged from an increase of 75 mm (Lettau et al. 1979) to reductions of about 200 á 230 mm (Henderson-Sellers & Gornitz 1984; Potter et al. 1975) or 100 to 800 mm (depending on location; Wilson (1984) in Henderson-Sellers 1987). Although part of these discrepancies arose from differences in the nature of the conversion (the studies by Wilson and Henderson-Sellers & Gornitz related to a conversion to grassland), it was clear that at least the results obtained for changes in temperature did not match observed values (cf. Schulz 1960; Pinker 1980; Lawson et al. 1981; Luvall 1984). Henderson-Sellers concluded that both land surface parameterisations (LSP) and global circulation models (GCM) were in need of improvement.

Since Henderson-Sellers her essay was written (late 1984/early 1985) two sophisticated LSP's, namely BATS ("Biosphere-Atmosphere Transfer Scheme; Dickinson 1984; Henderson-Sellers et al. 1986; Wilson et al. 1987) and SiB ("Simple Biosphere" model; Sellers et al. 1986; Dorman & Sellers 1989) have become available and calibrated for tropical rain forest (Dickinson (1989) and Sellers et al. (1989) respectively). The calibrations were carried out using the extensive micro-meteorological and plant physiological data set available for the Ducke forest reserve in central Amazonia (Shuttleworth 1988a; Roberts et al. 1990) and showed the strong superiority of the improved LSP's over earlier over-simplified ones (Shuttleworth et al. 1990).

The on-site soil-vegetation-atmosphere models (i.e. the LSP's) may be appended to the base of a number of grid areas within a GCM and by changing the LSP for forest to, say, degraded grassland, the climatic effects of the conversion may be simulated. Two such simulations have been carried out recently for the Amazon basin (Dickinson & Henderson-Sellers 1988; Shukla et al. 1990).

The former study concentrated on the on-site effects of changes in albedo and/or surface roughness (forests create more turbulence than a smoother surface like grassland) on foliar and soil temperatures, latent and sensible heat fluxes, "runoff" and soil water over a range of monthly rainfalls. Air temperatures following the conversion typically rose by 1-3 °K and soil temperatures by 2-5 °K (Dickinson & Henderson-Sellers 1988) which must be considered as a major improvement compared to the results quoted earlier in this section, even though the inferred warming of the surface is model dependent.

Dickinson & Henderson-Sellers (1988) also found reduced evaporation (up to 50 per cent) and precipitation (ca. 20 per cent) as well as a lengthening of the dry season after conversion. However, since the GCM they employed showed weaknesses in the simulation of the extent and duration of convective cloud cover (Shuttleworth et al. 1990), these results should be viewed with caution for the time being.

The successful calibration of SiB for a tropical rain forest (Sellers et al. 1989) opened the possibility for a more realistic estimate of the climatic effect of a large-scale conversion of Amazonian forest to grassland. Using a GCM of relatively high resolution (Kinter et al. 1988) and synthesising an LSP for degraded grassland from the literature, Shukla et al. (1990) carried out a simulation over a 13-month period. The result of the exercise is summarised in Figure 14.

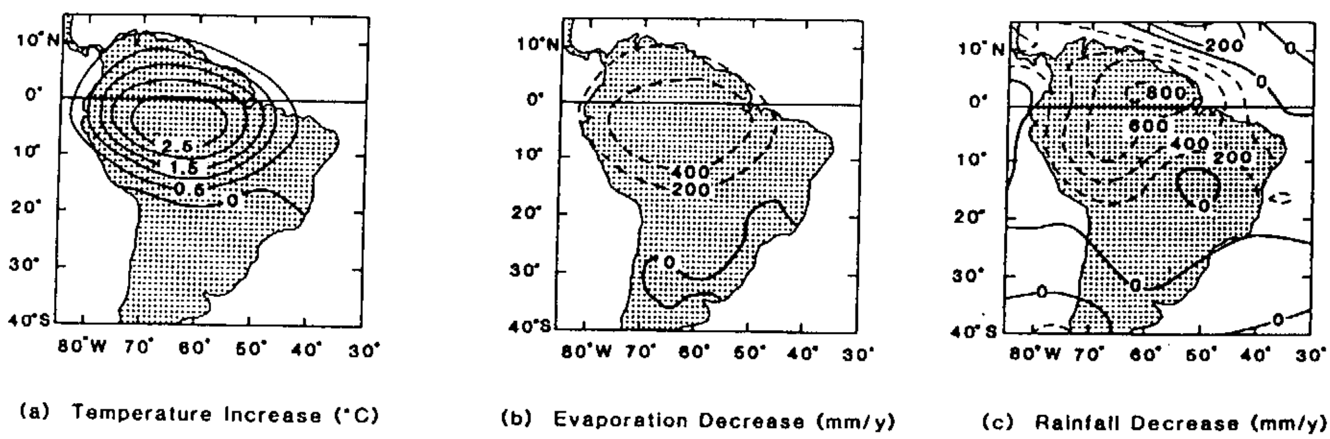

Figure 14. Predictions of the change in climate following a conversion of Amazonian rain forest to grassland (after Shukla et al. 1990).

Depending on location, predicted rises in air and soil temperatures were up to 2.5 and 3.5 °C respectively (as found by Dickinson & Henderson Sellers 1988), whereas basin wide rainfall and evapotranspiration were reduced by 26 and 30 per cent respectively (see Figure 14 for regional variations). The reduction in calculated annual precipitation by 640 mm and in ET by almost 500 mm suggests that changes in atmospheric circulation brought about by the conversion may act to further reduce the convergence of moisture flux in the Amazon region, a result that could not have been anticipated without the use of a dynamic model of the atmosphere (Shukla et al. 1990). For instance, Shuttleworth (1988a) predicted a drop in ET of about 20 per cent which was converted (on the basis of the recycling hypothesis; Salati et al. 1979) to a reduction in rainfall of about 10 per cent. Also, a reduction in ET could have been compensated for by an increase in moisture flux convergence.

The results of the experiment of Shukla et al. (1990) suggest that a compensation of this kind may not occur. Whether this is model-dependent will have to resolved by additional experiments and comparison with predictions by other models. If it is real, however, the ecological consequences will be enormous. As pointed out by Salati & Vose (1984) the presently experienced dry period in central Amazonia is the maximum the ecosystem can withstand. Any lengthening of the dry season will increase fire hazard (cf. Uhl & Buschbacher 1985) as well as influence a host of plant-animal interactions (Prance 1985, 1986b) which in turn could lead to irreversible changes in the vegetation.

Although the simulation by Shukla et al. (1990) is by far the most sophisticated of the attempts published to date, it should not be forgotten that the LSP for degraded grass- and scrubland employed in the study was based on data assembled from the literature rather than on actually determined values. Obviously, the use of other values for such important surface characteristics as reflection coefficient and soil water capacity might have produced different results (cf. Dickinson & Henderson-Sellers 1988). For example, in light of observations by Pinker et al. (1980) on the radiative properties of cleared areas occupied by grasses and scrubs in Thailand (13.4-14.3 per cent depending on season), the choice of a reflection coefficient of 21.6 per cent by Shukla et al. seems rather high. Similarly, Luvall (1984) reported net radiation totals over primary forest and a fresh clearing in Costa Rica to be very similar, although there was a profound change in the Bowen ratio (i.e. the partitioning of energy between sensible and latent heat). Therefore, the recently announced Anglo-Brazilian Amazonian Climate Observational Study (ABRACOS) which will study the water and energy dynamics of degraded Amazonian pastures (Shuttleworth et al. 1990) is a most welcome contribution in this respect and makes one look forward to its results. In addition, ABRACOS will address differences in micro-climate between large clearings and adjacent forest land in both the eastern (drier) and western (wetter) parts of Amazonia. In this way, it is hoped to improve the performance of GCM's in simulating extent and duration of convectional cloud cover as well as to improve the predictive ability of climate models in general (Shuttleworth et al. 1990; cf. Henderson-Sellers & Pitman 1990).

Summarising, great advances have been made in our understanding of forest-atmosphere interactions in the last few years. Although a number of questions remains to be answered (e.g. with respect to the maximum area that may be cleared without deleterious meso-scale climatic effects), it seems clear that the original contention of negligible influence put forward by Bernard (1945) and supported by Penman (1963) is no longer tenable in the light of more recent research results (Salati et al. 1979; Shukla et al. 1990).

4.3 Effect on water yield

Another common notion about the role of forests is that the complex of forest soils, roots and litter acts as a sponge soaking up water during rainy spells and releasing it evenly during dry periods. Although forest soils generally have higher infiltration and storage capacities than soils with less organic matter (Pritchett 1979), often much of this water is consumed again by the forest rather than used to sustain streamflow (cf. Table 1). Moreover, appreciable quantities of rainfall (up to 35 per cent; section 2.3.4) may be intercepted by the canopies of tropical forests and evaporated back into the atmosphere.

When dealing with the issue of effects of forest conversion on streamflow it is helpful to distinguish between effects on *water yield* (i.e. total streamflow) and on *flow regime* (the seasonal distribution of flow). The present section summarises what is known about the former whereas section 4.4 will discuss the latter aspect in more detail. Before examining the available information on effects of land use changes in the tropics on water yield a few methodological comments are necessary.

Simply comparing streamflow totals for catchment areas with contrasting land use types may lead to wrong conclusions because of the possibility of differences in catchment leakage (section 2.3.3). For example, Richardson (1982) found water yields for small catchments covered with montane rain forest and mature plantations of *Pinus caribaea* in Jamaica to differ by about 150 mm yr^{-1}. However, the corresponding values for ET were in excess of what has been reported for similar forests elsewhere by about 700 mm (Table 2), suggesting considerable catchment leakage. Similarly, flow from a 50-year-old plantation of *Eucalyptus robusta* in upland Madagascar was some 210 mm yr^{-1} below that for natural forest (Bailly et al. 1974; wet years only). Since the eucalypt covered catchment was much smaller than the rain forested one, and since small catchments in the area showed consistently lower streamflow totals than larger ones with the same vegetation (Bailly et al. 1974), it remains to be seen to what extent the quoted difference in flow reflects a real vegetation effect or rather a difference in catchment leakage.

Another complicating factor in the evaluation of hydrological effects of land cover transformations is the strong year to year variability of weather in the tropics (Qian 1983; Dyhr-Nielsen 1986). In addition, there is the large spatial variation in convective tropical rainfall which may render estimates of areal precipitation inputs for forested basins (which often have low rain gauge densities) relatively unreliable (Ribeny & Brown 1968; Aitken et al. 1972; De Bruin 1977).

An effective way to overcome some of these problems is the so-called "paired catchment method". Basically, the technique involves the hydrological comparison of two (or more) catchments of (preferably) similar size, geology, slopes, exposure and vegetation, situated close to one another: a "control" (to be left unchanged), and an "experimental" or "treatment" basin (Roche 1981; Hewlett & Fortson 1983).

The comparison is made during an initial calibration phase (which may

Plate 1. Scientific evidence is growing that evaporation from extensive tracts of rain forest produces part of its own rainfall.

Plate 2. "Cloud forest" at 2400 m on Mt. Kinabalu, Sabah, Malaysia.

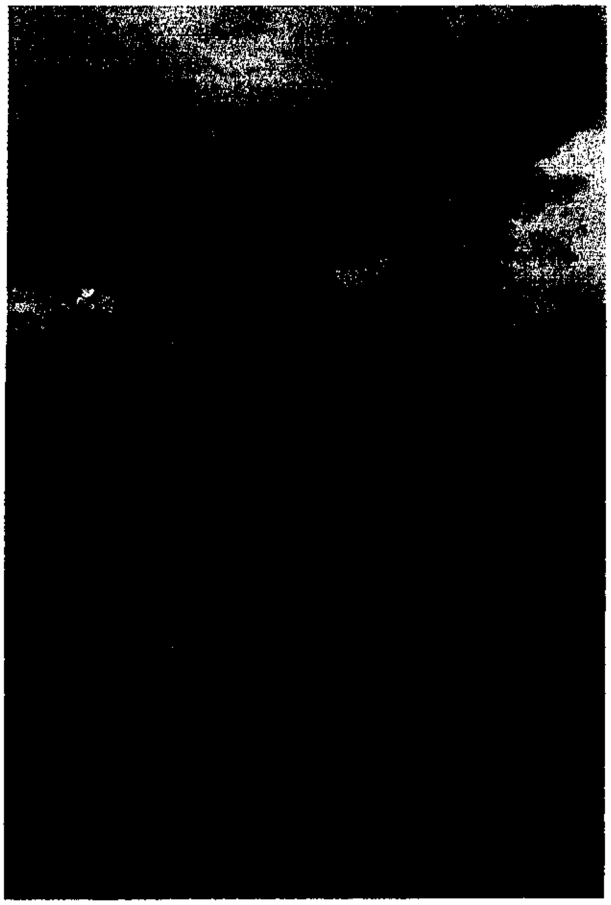

Plate 3.

Shifting cultivation in East Kalimantan, Indonesia (photo by K.F. Wiersum).

Plate 4. Rain forest that was selectively logged about fifteen years ago, Danum Valley, Sabah, Malaysia.

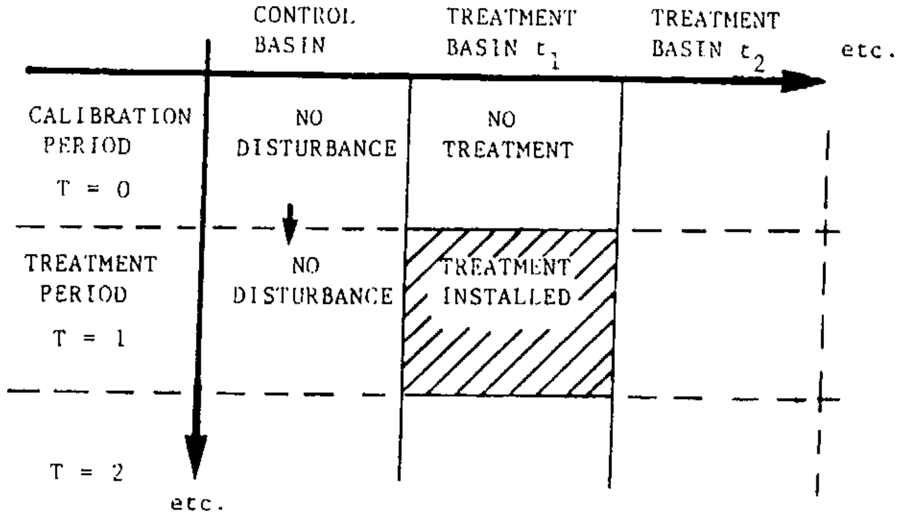

Figure 15. The paired catchment technique: general experimental design (after Hewlett & Fortson 1983).

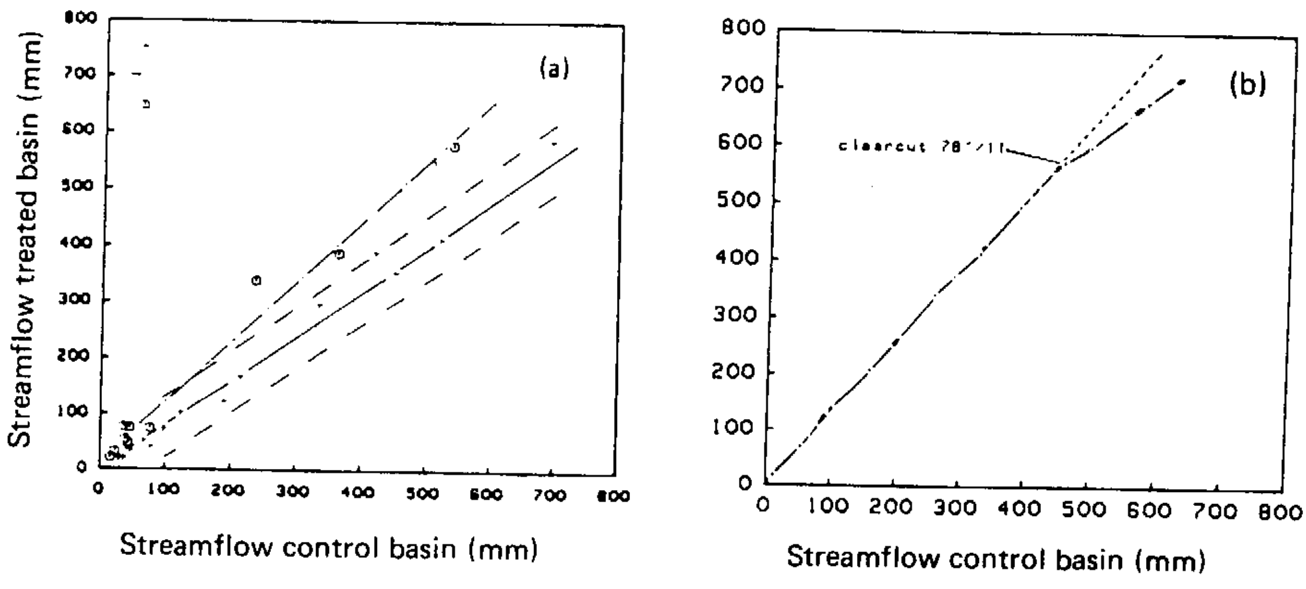

Figure 16. Evaluation of treatment effect by statistical analysis: (a) linear regression, (b) double mass curve (after Hsia & Koh 1983).

take several years, depending on rainfall variability (Kovner & Evans 1954), and during a subsequent treatment period in which the change in land use is effected (Figure 15). The degree to which linear regression equations (Figure 16a) or double mass curves (Figure 16b) linking the streamflows of the two catchments (as derived during the calibration period) change after the treatment is a measure of the effect of the latter.

The total duration of this type of experiment may easily span a decade in certain cases (calibration, clearing, site preparation, planting, maturation of the new vegetation; cf. footnotes Table 4). In addition, the results may be rather site specific due to an area's geological or pedological setting (Fritsch et al. 1987; Dano 1990). Therefore, in recent years there has been an increasing trend to predict hydrological changes brought about by land cover transformations in the tropics by robust models employing data obtained during relatively short (up to two years) but intensive measuring periods (Vugts & Bruijnzeel 1988; Shuttleworth 1988a; Shuttleworth et al. 1990; Institute of Hydrology 1990).

Bosch & Hewlett (1982) reviewed the results of almost hundred paired basin experiments throughout the world including a few from the tropics to determine the effect of vegetation removal or modification on basin water yield. They concluded that "no experiments in deliberately reducing vegetation cover caused reductions in water yield, nor have any deliberate increases in cover caused increases in yield".

In other words, *removal of forest cover leads to higher streamflow totals and reforestation of open lands generally leads to a decline in overall streamflow.*

Table 4 summarises the presently published data with respect to the effect of land cover transformation on water yield in the humid tropics, including a few examples from somewhat more seasonal subtropical sites. Most of the data are based on paired catchment experiments (see footnotes for details).

Results have been listed according to site elevation, starting with low intensity disturbances in lowland forests (no's 1-3), followed by various types of conversion in lowland areas (no's 4-7) and in more elevated terrain (no's 8-15). The last five studies (no's 16 - 20) deal with effects of reforestation, burning and coppicing.

The following conclusions may be drawn from Table 4:

(1) Carefully executed light selective harvesting will have little (if any) effect on streamflow (no's 2 and 9), whilst the effect increases with the amount of timber removed (case study no. 1);

(2) The data set for the humid tropics supports the general finding of Bosch & Hewlett (1982) that removal of the natural forest cover may result in a considerable initial increase in water yield (up to 800 mm yr^{-1}; possibly more in high rainfall regions: no. 4), depending mainly on the amount of rain received after the treatment (no's 1, 2, 5, 8, 12);

(3) Depending on rainfall patterns, there is a rather irregular decline in streamflow gain with time associated with the establishment of the new cover (no's 2, 5, 8); no data have been published regarding the number of years needed for a return to pre-cut streamflow totals in the case of natural regrowth; accepting the results of no. 11 at face value, this may take more than eight years; more work is needed;

(4) Water yield after maturation of the new vegetation may remain above original streamflow totals in the case of conversion to annual cropping (no's 7, 14), grassland (no's 4, 6, 12, 15 with larger increases under wetter conditions) or tea plantations (no. 13), return to original levels (pine plantation after full canopy closure: no. 14) or remain below previous values (reforestation of grassland with pines or eucalypts: no's 18-20); coppicing of eucalypts after ten years caused even stronger reductions for two years (no. 18);

(5) Burning of grassland may increase flow (no. 19, *Loudetia, Aristida*) or reduce it (no. 15, *Imperata*); in the former case this was mainly due to an increase in stormflow (burning every other year), in the latter it may be related to enhanced water uptake during and after renewal of above-ground biomass (annual burning); further investigation is desirable.

TABLE 4. Changes in annual water yield associated with changes in land cover in the (sub)tropics

Location	Type of transformation	Method	Change in water yield (mm yr^{-1}) 1st year	2nd year	3rd year	4th year	nth yr	
(1) Bukit Berembun, Malaysia[1-3]	Commercial selective logging (catchment C1, 40 % removal)	PCT	165^{w} (70%)	140 (53%)	175^{w} (72%)			
	Supervised selective logging (catchment C3, 33 % removal		85^{w} (37%)	70 (28%)	105^{w} (44%)			
(2) Babinda, Queensland[4]	Selective logging (N Creek)	PCT	effect not statistically significant					
	Clearing 67 % of basin and subsequent regeneration		265 (7%)	325^{w} (13.4%)				
(3) La Selva, Costa Rica[5]	Manual clearing (no burn) and subsequent regrowth	P-M	375^{*} (26%)					
(4) St. Elie, Fr. Guyana	Rain forest to eucalypt plantation[6-8]	PCT	$410/825^{*}$	$330/680^{*}$	$40/185^{*}$			
	Idem to <u>Pinus caribaea</u>[7,8]		$495/925^{*}$	$380/725^{*}$	$210/435^{*}$			
	Idem to <u>Digitaria</u> grass on poorly drained soil[9]		average over four years: $235/270^{*}$					
	Idem to <u>Brachiaria</u> grass on freely drained soil[9]		average over four years: $230/325^{*}$					
(5) Sungei Tekam, Malaya[10-15]	Rain forest to cocoa	PCT	110^{d} (117%)	706^{w} (157%)	353^{w} (94%)	263 (158%)		
	Rain forest to oil palm (60 %; remaining 40 % cleared in 3rd/4th yr		145 (85%)	155^{d} (142%)	137^{d} (97%)	822^{w} (470%)	793^{w} (270%)	476 (314%)

(6)	Yangambi, Congo[16]	Secondary forest versus	SWB				
		- Paspalum grassland					60* (14%)
		- Bare soil					285* (66%)
(7)	IITA, Nigeria[17,18]	Clearing secondary forest for traditional cropping	CWB	305*w	170*	140*	
(8)	Lien-Hua-Chi, Taiwan[19,20]	Clearing of mixed evergreen hill forest, regeneration cut in 2nd year	PCT	450 (58%)	205d (51%)		
(9)	Rajpur, India[21]	20 % thinning of Shorea forest	PCT	no detectable change in first two years			
(10)	Blue Mnts., Jamaica[22]	LMRF vs. mature Pinus caribaea plantation	CWB				150*
(11)	Périnet, Madagascar[23]	Montane forest vs. sec. bush	CWB				200*
		Idem vs. mature Eucalyptus					-210*
(12)	Luano, Zambia[24]	Miombo woodland to cropping (ca. 10%) and grazing (2/3)	PCT	average increase over five years: 195 - 230 (56 - 74 %)			
(13)	Kericho, Kenya[25]	Montane forest to tea (54%)	PCT				
		I clearing/planting		three-year average: 220			
		II establishment		four-year average : -100d			
		III maturation		six-year average : 150			
(14)	Kimakia, Kenya[26]	Montane forest (bamboo) to plantation of Pinus patula	CWB				
		I intercropping phase		three-year average:			125
		II establishment		six-year average :			75
		III maturation (closed canopy)		seven-year average:			5

1-25,*see footnotes; PCT = paired catchment technique; P-M = Penman-Monteith model; SWB, CWB = site, catchment water balance; w,d = wet, dry year or period

TABLE 4 continued

Location	Type of transformation	Method	Change in water yield ($mm\ yr^{-1}$) 1st year	2nd year	3rd year	4th year	nth yr
(15) Mbeya, Tanzania[27]	Montane forest (67%), scrub (33%) vs. traditional agriculture (50% crop, 50% grass)	CWB	ten-year average :				410*
(16) Angat, Philippines[28,29]	Imperata grassland to fire protected grassland	PCT	four-year average :				120 (9.5 %)
	Idem to Gmelina arborea		six-year average :				80 (9.4 %)
(17) Pidekso, Indonesia[30]	Degraded cropland vs. mixed plantation forest (11 yr)	CWB					-580
(18) Dehra Dun, India[31-34]	Scrubland to Eucalyptus	PCT	five-year average:				-15 (28 %)
	Coppicing of eucalypts after ten years		-160[d] (68%)	-285 (47%)	+10 (2%)		
(19) Manankazo, Madagascar[35]	Natural grassland to burned grassland	CWB/ SWB	11/6-yr-averages: 100 / 125				
	Idem to agriculture		11-yr average : - 75				
	Idem to Pinus patula		11-yr average : - 80 / 160 (6-yr-old trees)				
(20) Ootacamund, India[36]	Natural grassland to Eucalyptus globulus (59%)	PCT	average for first three years :				-10 (2 %)
			average between 4th and 10th yr:				-120 (21 %)

[26-36]see footnotes; SWB, CWB = site, catchment water balance; PCT = paired catchment technique; d = dry year

Table 4 continued (footnotes)

[1]Abdul Rahim & Baharuddin (1986); [2]Abdul Rahim (1989); [3]Abdul Rahim (1990); Experiment initiated in 1979; 3 adjacent rain forested basins at 170-300 m a.s.l.; deep Ultisols; granite; mean P ca. 1900 mm without real dry season; after 3 yr calibration against control basin (4.6 ha) commercial logging (40% stocking removed; 1.86 km of road/track - 0.14 km ha^{-1};) of basin C1 (13.3 ha) and supervised logging (30% removed; 2x20 m riparian buffer zone; 2.2 km road/track - 0.10 km ha^{-1}) of basin C3 (30.8 ha); crawler tractors and winch lorries.

[4]Gilmour (1977b);
Experiment initiated in 1969; mesophyll vine forest at 10-200 m; deep Ultisols with permeability rapidly decreasing with depth; metamorphic rock; wet with no mo with P <100 mm; after 18 mo of calibration against 25.7 ha control North creek basin (18.3 ha) selectively logged in 1971, 67% cleared and ploughed in 1973, then regeneration; area cleared again in December 1987 (M. Bonell, personal communication).

[5]Luvall (1984);
One-yr experiment; rain forest at 150-180 m; deep volcanic soils; P ca. 4000 mm without real dry season; ET via Penman-Monteith formula for primary forest and manually cleared 50x50 m block during 161 days; no removal or burning of slash; climatic data above-canopy except for wind; aerodynamic resistance computed from 10 day measurements of wind speeds at two levels above low vegetation; quoted value for 1st yr regrowth based on actual values for 1st half and taken as 80% of forest ET during 2nd half; since forest ET overestimated, results normalised to mean ET of Table 1.

[6]Fritsch (1983); [7]Fritsch & Sarrailh (1986); [8]J.M. Fritsch, personal communication; [9]Fritsch (1987); Large experiment initiated in 1977 lowland rain forest; soils with highly variable drainage; schists; climate wet and slightly seasonal; ten catchments (1-1.5 ha) calibrated for 17 (conversion to grassland) to 30 (conversion to plantations) mo before logging with heavy machinery; 1st values quoted refer to quickflow (Qq) and are given directly in [6-9], 2nd ones pertain to total runoff (Qt) computed from scattered data on Qq and Qt in [6-9] and are approximate; eucalypts (species not mentioned) exhibited very poor growth even after 2nd planting[8].

[10-11]DID (1975,1982,1986,1989); [14-15] Abdul Rahim (1987,1988); Experiment initiated in 1973, data available since 1977; lowland rain forest at 70 m; P ca. 1880 mm without real dry season; deep Oxisols on lateritic shales (control basin C, 56.2 ha) or on andesitic rocks and tuffs (experimental basins A and B); A (37.7 ha) headwater area of sub-B (59.2 ha), together making up B (96.9 ha); after 3 yr calibration sub-B (60% of B) logged and burned 2nd half '80; planting of cover crop spring '81 (= 1st yr); road construction and stream realignment in 2nd yr; planting of oil palm beginning of 3rd yr (late '82);

basin A (40% of B) logged and cleared between late '82 and mid '83 after 5 yr calibration; road construction and cover crop planting in 2nd half of '83 (= 2nd yr); cocoa planted late '83 and early '84, i.e. ca. 1 1/2 yr after planting of oil palm in sub-B.

[6]Focan & Fripiat (1953);
Secondary evergreen seasonal forest, Paspalum grassland and bare soil at 470 m; sandy clay soils; P ca. 1860 mm; seasonal climate but no details given; site water balance (cf. footnote 10 to Table 1 for details); values appear realistic.

[7]Lawson et al. (1981); [18]Lal (1983);
Experiment initiated in 1974; immature (?) secondary seasonal forest; Alfisol; P ca. 1450 mm with two dry seasons; CWB for single basin of 44 ha before and after clearing; no calibration; quoted values read from graph in [18] and 30 mm subtracted for flow in forested state[17]; values approximate.

[9]Hsia & Koh (1983); [20]Hsia (1987);
Study initiated in 1975; evergreen mixed broadleaf and coniferous forest at 725-785 m; fine silt loam soil; sandstone and shale; P 2100 mm, 80% between May and September; after 7 yr calibration against 5.9 ha control area, experimental basin (8.4 ha) cleared by skyline logging and uphill yarding; road around basin perimeter; minimal disturbance of soil; regrowth in 2nd yr cut for planting of China fir.

[1]Subba Rao et al. (1985);
Study initiated in 1964; dense Shorea forest at 895 m; deep sandy soils; P 2950 mm, mostly between June and October; flow not perennial; after 10 yr calibration against 6.5 ha control, experimental basin (5.2 ha) subjected to 20% thinning; measurements continued for 5 yr but only 1st 2 yr discussed.

[2]Richardson (1982);
One-year comparison of CWB for 38.5 ha basin with LMRF and 8.8 ha basin with 19-yr-old pines; no calibration; P 3750 mm with no real dry season; gravelly sandy loam; porphyrites and conglomerates; very high ET probably due to catchment leakage; quoted difference not reliable.

[3]Bailly et al. (1974);
Experiment initiated in 1962; long-term comparison of CWB for 38 ha basin with evergreen montane forest (D3) with that for 31.5 ha basin with secondary scrub (D7) and for 13.3 ha basin (D5) with 50-yr-old eucalypts; forest selectively logged 50 yr ago, plantation thinned just before start of experiment; vigorous undergrowth in D5; 930-1095 m; Ultisols; gneiss; P variable (1600-2100 mm) with 7 mo P<100 mm (2 <60 mm); all basins had excellent instrumentation; P at D5 some 500 mm below that for D7 and D3; quoted difference in Q based on average runoff coeffients and P of 2000 mm; difference between forest and scrub seems high but feasible, that between forest and eucalypts may be affected by leakage from D5.

[24]Mumeka (1986);
Experiment initiated in 1964; mixed Miombo (semi-open) woodland on freely drained soil surrounding seasonally inundated grasslan (dambo) on poorly drained soil; 1300 m; P 1400 mm; strongly seasonal (no details given); three excellently instrumented basins (95-143 ha) intercalibrated for 8 yr; 2 basins (A and G) cleared except for 60 m wide buffer strip around dambo; 10 ha used for improved agriculture, rest used as pasture.

[25]Blackie (1979a);
Experiment initiated in 1958; montane rain forest at 2200 m; dee volcanic soils; phonolitic lava; P 2130 mm with two months P<100 mm; after 18 mo calibration against 544 ha control basin (Lagan) 54 % of Sambret basin (702 ha) was cleared to establish tea plan ation; see footnotes Table 2 for instrumental details.

[26]Blackie (1979b);
Experiment initiated in 1958; montane rain forest at 2440 m; dee volcanic soils; pyroclasts over basalts weathered into dense cla P 2300 mm; CWB for forested basin C (64.9 ha) and for 36.4 ha ba A planted with pines just before observations; no calibration; basins believed watertight; see footnotes Table 2 for instrument details.

[27]Edwards (1979);
Experiment initiated in 1958; montane evergreen forest at 2500 m; one metre of volcanic soil over deeply weathered gneiss; P 1925 r with 6 mo with P<60 mm (4 completely dry); CWB for forested catch ment (C) of 16.3 ha and agricultural basin (A) of 20.2 ha; no cal ibration and some leakage (cf. high ET forest in Table 2) cannot excluded; no soil conservation measures on cropland in A; see footnote Table 2 for instrumental details.

[28]Dano (1990); [29]A.M. Dano, personal communication;
Experiment initiated in 1973; <u>Imperata</u> grassland at 225 m; clay loam soils; andesitic rock?; P 3170 mm with 3 mo P <60 mm; after 4 yr calibration against 1.6 ha control, Left Fork basin (0.95 ha protected against annual burning for 4 yr (1977-1980), then plant with <u>Gmelina</u> (poor growth, data until 1986[29]); burning continued control; only small part of Qt recorded which may have affected quoted values; highest increases during dry years and vice versa.

[30]Hardjono (1980);
One-yr comparison of CWB for 354 ha basin with mixed 11-yr-old pine, teak and mahogany plantations and 207 ha basin with dryland agriculture; seriously degraded volcanic terrain; 375 m; thin soils; P 2600 mm with 4 mo of P <60 mm; no calibration; quoted values most probably affected by differences in basin leakage (cf. section 4.4.2).

[31]Mathur et al. (1976); [32]Mathur & Sajwan (1978); [33,34]Vishwanatham et al. (1980, 1982);
Experiment initiated in 1961; secondary scrub (5 yr old at start of observations) at ca. 520 m; stabilised gullies in undulating terrain; silty (clay) loam soils; P ca. 1430 mm, mostly between June and October; flow not perennial; quoted data refer to Qq; after 8 yr calibration against 0.87 ha control, basin 2 (1.45 ha) planted with Eucalyptus grandis and E. camaldulensis (2x2 m); results given as means for 5 yr after treatment; post-treatment included 2 very dry years.

[35]Bailly et al. (1974);
Experiment initiated in 1962; long-term comparison of CWB for 3.2 ha basin with natural grassland with that for 4.8 ha basin with grassland burned every other year, for 3.2 ha basin with improved dry-land agriculture and for 3.9 ha basin with newly planted pine; 1550 m; Ultisols; gneiss; P 1715 mm with 7 mo of P <60 mm; values refer to Qq.

[36]Samraj et al. (1988);
Experiment initiated in 1968; natural grassland at 2035 m with scattered stunted evergreen trees and swampy valley bottoms; deep permeable soils; P 1535 mm, concentrated between June and September; after 4 yr calibration against 33.2 ha control basin, 59% of catchment B (31.9 ha) planted with bluegum (2x2 m); trees grown in 10-yr rotation.

The almost complete lack of rigorous (i.e. involving basin calibration) studies dealing with the impacts of shifting cultivation or conversion of natural forest to annual cropping on streamflow is somewhat surprising since these two activities account for a major portion of tropical forest destruction (Myers 1980; Seiler & Crutzen 1980). Virtually all material deals with effects on infiltration capacity, surface runoff and on-site erosion (see section 4.5).

Bailly et al. (1974) compared amounts of flow emanating from small (1.4-1.8 ha) basins in upland Madagascar under old secondary vegetation, dry land agriculture with soil conservation measures and under temporary slash and burn agriculture (two years followed by regeneration), respectively. Since the small basins were leaking severely, results pertained to stormflow only. Runoff was consistently highest from the third basin and lowest from the first one. However, it is difficult to envisage how runoff from six-year-old regrowth may be more than twice as high as for old scrubland as reported by Bailly et al. (1974). Again, it cannot be excluded that differences in catchment characteristics have influenced the results (see also section 4.4.1).

Changes in flow from a catchment in Nigeria (no. 7 in Table 4) whose secondary forest cover was converted to dry land agriculture as reported by Lal (1983) were a little contradictory. On the one hand, surface runoff gradually increased with time after clearing, supposedly as a result of deteriorating infiltration opportunities. The associated decrease in soil water storage would have led to reduced baseflows but instead these were reported to increase over the three year period (Lal 1983). Since the experiment did not include a calibration of the catchment it is likely that the reported trends were influenced by variations in rainfall.

Edwards (1979) presented a long-term comparison of streamflow from two small and presumably watertight basins in upland Tanzania, one with evergreen forest and one converted to (traditional) agriculture just before the start of the observations (no. 15 in Table 4). A consistent difference in water yield of about 400 mm yr^{-1} was observed during the ten year period of measurement. Incidently, throughout the entire duration of the experiment no environmental deterioration occurred in the cleared catchment, despite the fact that no soil conservation measures other than the bunding of harvest residues were applied, much to the surprise of the investigator himself (Edwards 1979). This rather unexpected result could be explained by a combination of relatively stable volcanic soils capable of maintaining high infiltration capacities (cf. Lungren 1980) and low intensity rainfalls.

Although circumstances at Mbeya are atypical for the humid tropics and these results must therefore be considered exceptional, they nevertheless illustrate the potential for watershed management in the tropics through modification of the surface cover *in conjunction with soil conservation measures* (cf. Bailly et al. 1974). Much is expected from agroforestry in this respect (Nair 1984; Ewel 1986; Young 1986; Vandermeer 1989; Bonell 1989) although little information is available as yet about the water dynamics of the various agroforestry systems.

Imbach et al. (1989) computed water balances for two such systems in Costa Rica, viz. cocoa with *Erythrina poeppigiana* and *Cordia alliodora* as the respective shade trees, using a somewhat insensitive method for the evaluation of transpiration and hence percolation. Taking their estimates of ET (800 and 1025 mm yr^{-1} respectively) at face value, permanent gains in water yield after conversion of upland forest to these types of agroforestry of about 200 and 400 mm could be expected (cf. Table 2).

Large tracts of lowland forest have been (and are still being) converted to extractive tree plantations such as oil palm and rubber in such rapidly developing countries as Nigeria, Ivory Coast or Malaysia (Bertrand 1983; Abdul Rahim 1985; Lal 1987) before the long-term hydrological consequences were known very well. As indicated before, paired catchment experiments studying the effects of conversion of tropical forest to tall vegetation take a long time to complete. Therefore, most of the studies quoted in Table 4 have covered only the early phase of such a conversion.

To answer the question whether water consumption of fast growing and regularly fertilised extractive tree crops in their mature phase will exceed that of the natural forest they have replaced, requires additional information.

As for oil palm, lysimetric studies on a young tree by Ling (1979) in Malaysia confirmed the initial observations of the paired catchment study at Sungei Tekam (no. 5 in Table 4), i.e. a reduction in water use compared to rain forest of 200-250 mm yr^{-1}. Later work by Foong et al. (1983) using the same large lysimeter showed an increase in annual ET to about iety of hydrological techniques, not all of them equally sophisticated, for areas ranging in size from small plots through catchment areas of several hectares to large drainage basins, thereby introducing problems of spatial representativity and extrapolation (Fritsch et al. 1987) as well as of comparability (Vitousek & Sanford 1986). In addition, observations have been made for various time scales (sometimes even less than a year), which introduces the problem of climatic variability (wet and dry years; Franken & Leopoldo 1984; Likens et al. 1977; Poels 1987).

The precise estimation of amounts of water draining from a soil is notoriously difficult (Cooper 1979, 1980). Some have approached the problem by trying to solve the *water balance equation* (Ward 1975) for the drainage component (Steinhardt 1979; Russell 1983; Parker 1985; Crowther 1987b). However, this technique will only yield reasonable values if evapotranspiration (ET) from the forest is known with a fair degree of accuracy and, as we have seen in Chapter 2, good estimates of tropical forest ET are rare. In addition, with this approach there is the problem of defining average nutrient concentrations of soil water (see below).

The classical *Darcyan approach* involves the measuring of soil water potentials below the root zone and the determination of the unsaturated hydraulic conductivity of the soil (Cooper 1979). Besides being laborious, the method may easily produce results that differ an order of magnitude from values obtained via other techniques, partly because of spatial variations in soil structure and hydraulic conductivity (Cooper 1979; Russell & Ewel 1985; Brugge 1988). Also, with well-structured soils there is the difficulty of preferential flow through macropores during heavy rains as opposed to matrix flow at other times (Beven & Germann 1982). The two flow types need to be dealt with separately, also in terms of their chemical composition (Haines et al. 1982; Bruijnzeel 1983b; Russell & Ewel 1985; Nortcliff & Thornes 1989).

So-called *"zero-tension lysimeters"* (Figure 11a; Jordan 1968) only sample the freely draining fraction of the soil water, the rest being held by capillary forces in the soil surrounding the lysimeter (Horton & Hawkins 1965). Zero-tension lysimeters therefore tend to underestimate downward flow (Radulovich & Sollins 1987), although occasionally the trapping of channelised subsurface flow may lead to the collection of volumes of water that are in excess of incident rainfall (Jordan 1969;

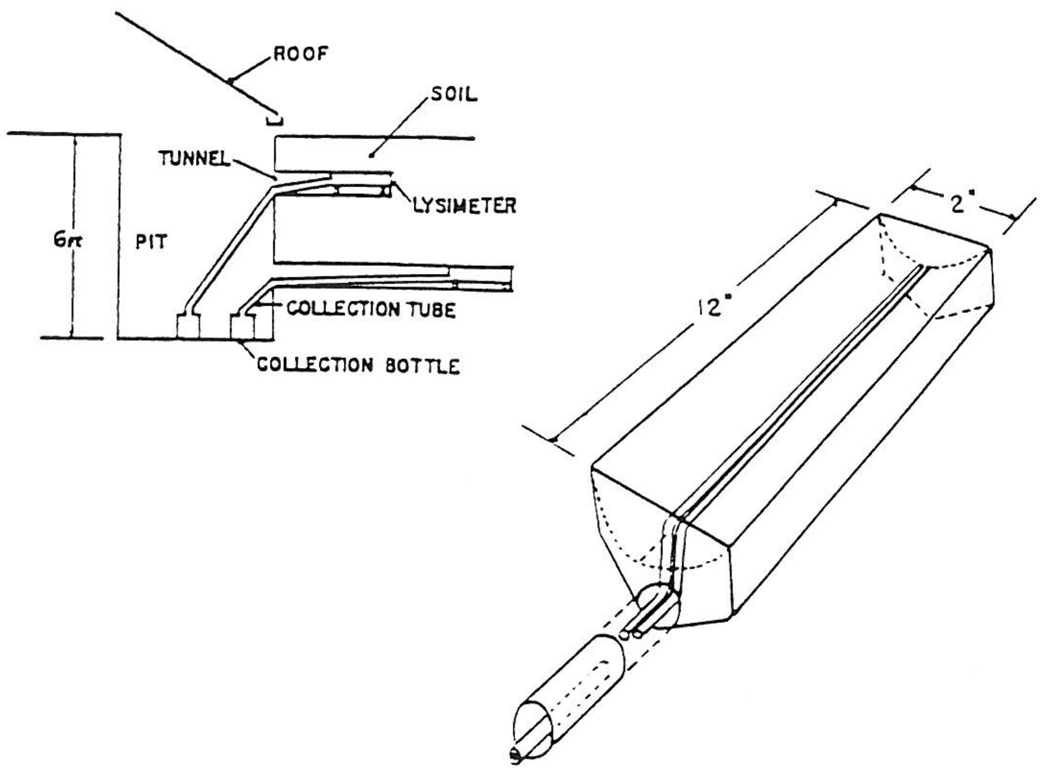

Figure 11a. Zero-tension lysimeter (after Jordan 1968).

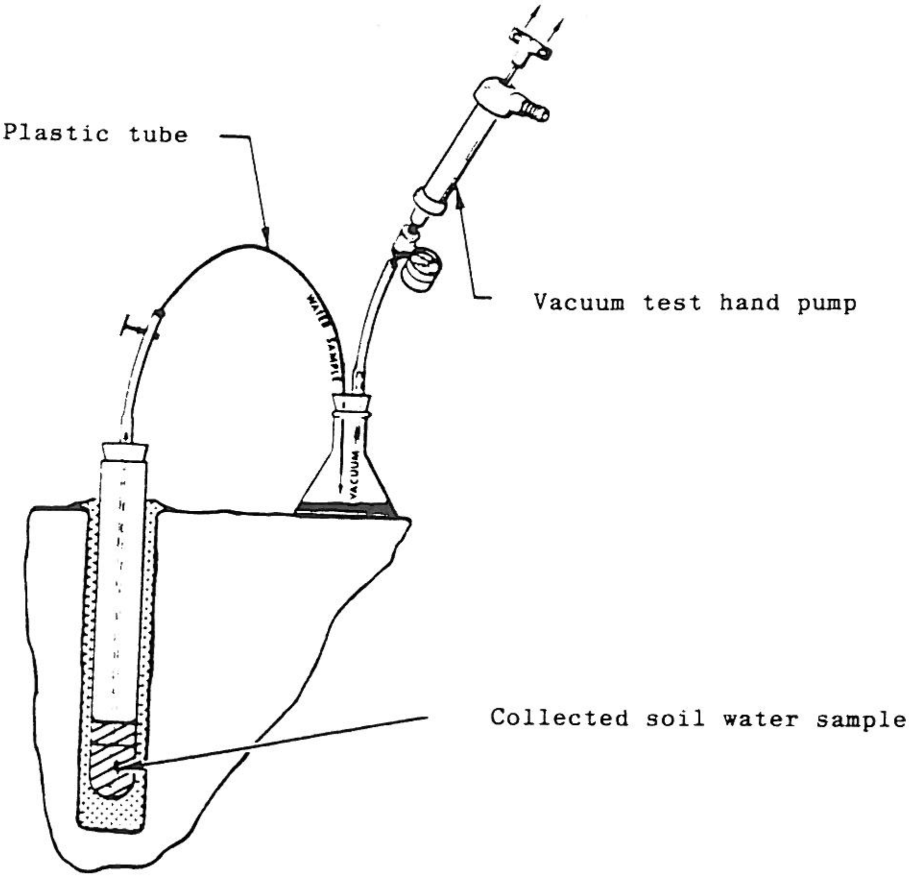

Figure 11b. Vacuum tube lysimeter for sampling of soil water.

Russell & Ewel 1985). Also, water sampled from zero-tension installations often exhibits nutrient concentrations that differ from those obtained with suction lysimetry (cf. Figure 11b), albeit not necessarily in a consistent manner (Haines et al. 1982; Russell & Ewel 1985). Finally, the composition of soil water sampled by suction lysimetry will depend on the magnitude of the suction applied (Nortcliff & Thornes 1989) as well as on the retention characteristics of the lysimeter cup or plate (e.g. ceramic or teflon; Zimmermann et al. 1978).

The use of a *catchment area* as the unit of measurement would eliminate these problems since streamflow integrates the various flow types, but may in some cases introduce problems of spatial variation in soil types, drainage situation and vegetation types (Lescure & Boulet 1985). For example, in parts of the Amazon one finds tall forest on well-drained Oxisols on higher grounds away from the streams ("Tierra firme") and an entirely different type of vegetation ("Caatinga") on the waterlogged sandy deposits in the valleys (Brünig et al. 1977). Clearly, in such cases one cannot expect the chemical composition of stream water to be representative of water draining from the Oxisols (Nortcliff & Thornes 1978,1989; Brinkmann 1985). Also, as commented upon in the preceding chapter, many catchments, especially small ones in headwater areas, are leaky and part of the flow may go unrecorded as deep leakage via rock fissures and fault zones or through the valley fill, thereby producing an underestimate of hydrologic nutrient outputs. Finally, as will be illustrated in the next section, nutrient losses from oligotrophic systems as determined by the catchment technique may or may not be representative of the losses experienced by the biologically active portion of the ecosystem, depending on the depth of the rock weathering front (releasing nutrients) with respect to that of the root network (taking up nutrients) and the degree of river incision (Baillie 1989, Bruijnzeel 1989b).

3.3.2 Computation of atmospheric nutrient inputs

It is difficult to obtain a realistic areal estimate of the amounts of nutrients entering a tropical high forest through wet and dry deposition (White & Turner 1970; Crozat 1979; Lewis et al. 1987), especially for nutrients with a gaseous phase (Delmas & Servant 1983: Servant et al. 1984). Atmospheric nutrient inputs to forest ecosystems in the tropics are traditionally approximated by multiplying periodictotals of rain by the corresponding nutrient concentrations (Parker 1983). Precipitation is usually measured and sampled in a forest clearing and sometimes above the canopy (Jordan 1969; Manokaran 1980). In this way an estimate of the nutrient content of bulk precipitation (an integration of wet and dry depostion, Whitehead & Feth 1964) is obtained, at least for the precipitation collector itself. For a number of reasons this estimate is not necessarily the same as the amounts of nutrients deposited on the forest canopy (White & Turner 1970). The latter is not only more exposed to the air passing overhead, but also represents a far larger (per unit area) trapping surface for aerosols, dust, etc. than a standard collecting funnel placed in a sheltered clearing (Mayer & Uhlrich 1974). The problem is confounded further by the observations of Crozat (1979), Delmas et al. (1980), Harriss (1987) and others which suggest that extended rain forest areas may generate their own aerosols.

Additional problems relate to spatial variations in rainfall inputs (by no means easy to quantify in tropical forest areas: Aitken et al. 1972; Lewis 1981) and enrichment of samples in the collectors by bird droppings or insect frass, litter, road dust, etc. (Kenworthy 1971; Galloway & Likens 1978; Poels 1987: Zulkifli Yusop et al. 1989).

3.3.3 Analytical problems

Undisturbed tropical forests are often found in remote places where it is difficult to maintain high standards for measuring hydrological variables and/or treatment and storage of water samples (Herrera 1979; Galloway et al. 1982; Jordan 1982; Poels 1987). However, it is important to remember that the comparability of results obtained at different sites is not only determined by the choice of hydrological techniques or the rigidity of collection and storage procedures for water samples (Ridder et al. 1985), but also by the selected analytical methods. For example, it is not always clear from a publication whether "N" or "P" indeed represent total quantities of nitrogen or phosphorus. Dissolved organic fractions (often not determined) may well constitute the bulk of total phosphorus or nitrogen in a tropical forest stream (Brinkmann 1985: Lewis 1986). A related and often overlooked aspect is that of the standard of laboratories (La Bastide & van Goor 1978).

Not surprisingly in the light of the above considerations Vitousek & Sanford (1986) concluded that no useful patterns with respect to nutrient losses from tropical forests could be drawn from the (rather limited: n=9) data set presented in their review. Rather, according to Vitousek & Sanford, the comparability of results should be improved first by using standardized (and partly novel) techniques. Although quite valid from an academic point of view, such a standpoint effectively means that it may well take a decade from now before sound comparisons can be made. An attempt at finding any patterns is made below by critically examining a somewhat larger data set (n=25) than that used by either Vitousek & Sanford (1986) or Proctor (1987).

3.4 Nutrient budgets for tropical forest ecosystems

3.4.1 General comments

When examining inputs, outputs and net losses of nutrients for tropical forest ecosystems, it is helpful to group the sites according to the assumed fertility of their substrates (Vitousek & Sanford 1986). Using the US Soil Taxonomy System (USDA 1975), a very low level of fertility may be assigned to soils classified as Spodosols (podzols) and Psamments (sandy soils lacking distinct horizonation), which often occur on bleached oligotrophic quartz sands. Low fertility can also be assumed for Histosols (peat soils) and all Lithic (shallow and stony) subgroups. Oxisols (thoroughly weathered "latosols") and Ultisols (less-weathered soils, including red-yellow podzolic soils) exhibit levels of fertility ranging from very low to moderately low, and are especially widespread in the lowland tropics. Conversely, Inceptisols (relatively young and unweathered "brown forest soils"), Alfisols (podzolic soils), Vertisols (black clay soils), Rendolls (rendzinas) and Fluvents (alluvial soils) constitute the bulk of a "moderately fertile" group (Sanchez 1976; Vitousek & Sanford 1986).

Table 3 presents atmospheric inputs and hydrologic outputs of macronutrients (calcium, magnesium, potassium, phosphorus and nitrogen) for some twenty five (sub)tropical forest ecosystems on a variety of geological substrates. Two of these budgets (no's 20 and 21) were compiled by the present writer by combining partial information for a particular area

TABLE 3. Additions in bulk precipitation (I), losses in drainage water (L) and the diffe

Location	Annual rainfall (mm)	Annual runoff (mm)	Calcium I	Calcium L	Calcium I-L
			($kg\ ha^{-1}\ yr^{-1}$)		
			LOWLAND FORESTS		
(I) Spodosols/Psamments					
(1)San Carlos, Venezuela[1,2*+]	3565	1860[a]	16.0	21.2	- 5.2
		1860[b]		18.9	- 2.9
		1860[c]		2.8	+13.2
Alternate computation[d]			5.2	2.8	+ 2.4
(II) Oxisols/Ultisols					
(2)Caura River, Venezuela[3-4*]	3850	2425	1.3	15.5	-14.2
(3)San Carlos, Venezuela[5-7+]	3565	1595	10.3	3.3	+ 7.0
Alternate computation[e]	3565	2100	5.2	5.6	- 0.4
(4)Tai Lam Chung, Hong Kong[8*]	1900	1235	7	18	-11
(5)Jari, Brazil[9+]	2300	1225	15.8	16.8	- 1.0
(6)Adiopodoumé, Ivory Coast[10+,*]	2130	1000	37.8	46.6	- 8.8
		1000[f]		21.1	+16.7
(7)Ulu Gombak, Malaysia[11*]	2500	750	14.0	2.1	+11.9
(8)Banco (valley site), Ivory Coast[12,13+]	1800	630	30	43	-13
		630[g]		10.3	+19.7
(9)Tonka, Surinam[14*]	2145	515	16.2	7.6	+ 8.6
(10)Ducke Reserve, Brazil[15,16*]	2475	ca.450	0.3	0.9	- 0.6
(11)Ibidem[17*]	2075	400	tr.	tr.	?
(12)Bt.Berembun, Malaysia[18-20*]	2005	225	41.8	5.8	+36.0
Alternate computation[m]	2000	575	3.6	14.8	-11.2

*catchment based study
[a-m]see footnotes next pages
+(zero-)tension lysimetry
°total soluble amounts unless indicated otherwise
▲excluding fixation inputs and denitrification losses

feren (I-L) for calcium, magnesium, potassium, phosphorus and nitrogen in (sub)tropical forest:

Magnesium			Potassium			Phosphorus			Nitrogen		
I	L	I-L	I	L	I-L	I	L°	I-L	I	L	I-L▲
				(kg ha^{-1} yr^{-1})							
Soils of moderate to very low fertility											
3.1	4.4	-1.3	18.0	26.0	-8.0	16.7	21.2	-4.5	21.2	15.3	+5.9
	2.2	+0.9		2.6	+15.4		3.6	+13.1		10.2	+11.0
	0.6	+2.5		3.5	+14.5		16.0	+0.7		8.2	+13.0
0.7	0.6	+0.1	6.9	3.5	+ 3.4				2.3	8.2	- 5.9
0.3	6.0	-5.7	1.0	14.6	-13.6	0.14	0.24	-0.1	(2.3)[2]	6.3	(4.0)
3.5	0.7	+2.8	12.6	4.6	+8.0	23.4	28.9	-5.5	11.5	14.1	-2.6
0.7	1.5	-0.8	6.9	7.0	-0.1	0.45	0.15	+0.3	2.3	10	-7.7
6.5	6	+0.5	7	18	-11						
3.3	8.1	-4.8	10.2	12.7	-2.5	0.14	0.04	+0.10			
8.4	30.4	-22.0	6.3	69.1	-62.8	2.2	2.1	+0.1	29.4	29.5	-0.1
	4.4	+4.0		0.9	+5.4		11.0	-8.8		14.1	+15.3
3.2	1.5	+1.7	12.5	11.2	+1.3						
7	4.3	+2.7	5.5	1.3	+4.2	2.3	0.2	+2.1	21.2	12.6	+8.6
	6.2	+0.8		2.8	+2.8[g]	0.1	+2.2			6.7	+14.5
2.4	4.4	-2.0	14.4	2.4	+12.0	0.8	0.3	+0.5	18.8[h]	0.15[h]	18.6[h]
0.2	0.5	-0.3				0.4	0.3	+0.1	5.0	29.0	-24.0
tr.	tr.	?	2.1	0.4	+1.7	0.104[h]	0.008[h]	+0.1[i]	6.0[j]	0.2[j]	+5.8[j]
16.9	3.6	+13.3	9.1	8.0	+0.9	0.6[k]	0.2[k]	+0.4[k]	3.9[l]	0.6[l]	+3.31
0.6	9.2	-8.6	6.8	20.5	-13.7						

rera (1979); [2]Buschbacher (1984)
aatinga" forest; highly leached and frequently water-logged sands; "black ater" area; amount of drainage computed by subtracting estimated evaporation total for Rio Negro river basin (126760 km^2) from average rainfall at San Carlos; nutrient concentrations of drainage water approximated by [a]weekly collections of 15 zero-tension lysimeters at -12 cm during October and November 1975, [b]groundwater at -100 cm (no specification of time period) and [c]streamwater (August 1975 - December 1976); nitrogen comprises inorganic fractions only; rainfall sampled between November 1975 and December 1976; [d]difference between atmospheric inputs as estimated by Buschbacher (1984) and streamwater outputs as before.

[3]Lewis (1986); [4]Lewis et al. (1987)
Undisturbed lowland forest; highly depleted soils on Precambrian rocks; spodosols in depressions; "black water" river; catchment area 47,500 km^2; bi-weekly depth integrated sampling of river water at four points across the stream between May 1982 and April 1984; nutrient inputs estimated via bulk precipitation inputs at basin outlet and adjusted on the basis of basin equilibrium for sulphur and chlorine in/outputs; dry-season contributions not included; high standards for storage and analysis of water samples.

[5]Jordan (1982); [6]Jordan & Heuveldop (1981); [7]Jordan et al. (1982); [2]Buschbacher (1984)
"Tierra firme" forest; highly leached oxisols; amount of drainage computed by subtracting "class A" pan evaporation from rainfall; average nutrient concentrations in drainage water as collected by 24 zero-tension lysimeters was computed by weighting samples obtained from lysimeters at -12 cm four times as heavily as concentrations at -40 cm to account for subsurface lateral flow; precipitation sampled from 20 polypropylene bottles with their orifices covered by a thin layer of glass wool; quoted data concern 1976-1979 annual totals; phosphorus as phosphate only; nitrogen comprises inorganic fractions; [e]inputs from Buschbacher (1984); outputs computed by subtracting average lowland rain forest ET (1425 mm yr^{-1}, Bruijnzeel 1989b) from rainfall to obtain drainage rate multiplied with average concentrations for lysimeters at -17 and -30 cm in a depleted ultisol as derived from unpublished data over 2.5 year by Buschbacher in Medina & Cuevas (1989).

[8]Lam (1978)
Dense natural regrowth of Pinus massoniana; deeply weathered granite; red-yellow podzolic soils; catchment area 24 ha; observation period July 1971 - October 1972 and wetter than usual; vegetation still acquiring biomass; all values approximate as they were derived from a graph and converted to annual values by multiplication with 0.8.

[9]Russell (1983)
Lowland rain forest; sandy ultisol low in calcium, potassium and phosphorus; drainage approximated as 50% of rainfall (Molion, 1975); corresponding nutrient concentrations by sampling six (?) zero-tension lysimeters at -30 cm during three periods (9 samples) in 1981; rainfall sampled daily at five stations between January 1981 and January 1982 using the approach of Jordan (1982) at San Carlos, Venezuela.

[10]Roose (1981)
Old secondary and slightly disturbed "evergreen" forest; Tertiary sandy and clayey deposits; oxisols; drainage estimated by a modified Thornthwaite soil water budget method (1964-1975); outflow from cylindrical (diameter 63 cm, height up to 120 cm) soil monoliths frequently sampled for chemical analysis and concentrations averaged for surficial and deeper layers; [f]outflow obtained by multiplying average spring water composition with average drainage rate.

[11]Kenworthy (1971)
Partly disturbed Dipterocarp forest; granite; oxisols; catchment area 31 ha; observation period August 1968 - February 1969 with measurements extrapolated to a one-year period; no details given for collection, storage, and analytical procedures; nutrient concentrations in rainwater rather high (Manokaran 1980), suggesting sample enrichment cannot be ruled out; streamflow probably underestimated, possibly because of leaky conditions.

[12]Huttel (1975); [13]Bernhard-Reversat (1975)
Environment as described for [10]; drainage computed via soil water balance with variations in soil water storage measured by neutron probe technique; during wet periods (net downward flow) drainage derived by subtracting forest ET (assumed equal to that computed with the Turc formula) from rainfall; observation period (1969-1971) received below average rainfall totals; two zero-tension lysimeters at -40 cm sampled weekly and analyzed after pooling to monthly samples; rainfall chemistry (1970-1972) approximated by that recorded at nearby Adiopodoumé; [g]nutrient outflow obtained by multiplying average streamwater composition with average drainage rate.

[14]Poels (1987)
"High dryland forest" on slopes and ridges, "marsh/swamp" forest in depressions; sandy loam deposits on granitic (?) Precambrian basement; well-drained yellowish brown oxisols on slopes/ridges, poorly drained grey sands (tropaquents) in valleys; catchment area 140 ha; observation period November 1979 - April 1984; weekly sampling of rainfall and streamflow; rainfall sampled from groundlevel gauge; samples with electrical conductivity > 50 microS/cm discarded; numerous other corrections applied to account for rather weak analytical facilities in Surinam; [h]inorganic fractions based on limited number of samples.

[15]Brinkmann (1983); [16]Brinkmann (1985)
"Terra firme" rainforest with "riverine" forest along streams; Tertiary sedimentary rock, depleted yellow oxisols on plateau and slopes; sandy hydromorphic soils in valleys; several small catchments and plots; observation period 1968-1972.

[17]Franken & Leopoldo (1984)
Vegetation and soils as [15,16]; catchment area 1.3 km^2; observation period September 1976 to September 1977 and rather dry; rainfall and streamflow sampled weekly; concentrations of calcium and magnesium generally within methodological error; [i]phosphate-phosphorus; [j]ammoniacal nitrogen.

Table 3 continued

Location	Annual rainfall (mm)	Annual runoff (mm)	Calcium I	L	I-L (kg ha^{-1} yr^{-1})
			Soils of moderate to high fertility		
(III) Inceptisols/Mollisols/Vertisols					
(13) Watubelah, Indonesia[21-23*]	4670	3590	9.9	29.0	-19.1
Alternate computation[q]				86.3	-76.4
(14) La Selva, Costa Rica[24+]	3675[r]	1250[r]	2.7	5.7	- 3.0
Alternate computation[u*]	4007	2580	3.1	9.2	- 6.1
(15) Kinta Valley, Malaysia[26,25*]	2845	1605	11.4	795	-784
				651[v]	-640[v]
(16) Ei Creek, Papua[27*]	2700	1480	0[w]	24.8	-24.8
(17) Gua Anak Takun, Malaysia[25,26+*]	2440	1255	36.1[x]	764	-728
				583[v]	-547
(18) Lien-Hua-Chi, Taiwan[28*]	2375	840	3.5	20.3	-16.8
(19) Darien, Panama[29*]	1935	855	29	163	-134
			MONTANE FORESTS		
Inceptisols/Ultisols					
(20) Rio Espiritu Santo, Puerto Rico[30-33*]	4550	4260	23	166	-143
(21) Mt. Kerigomna, Papua[34,35*]	3800	3160	3.6	(63)	(-59)
(22) El Verde, Puerto Rico[36,37*]	3920[A]	2150[A]	21.8	43.1	-21.3
(23) Pi-Lu-Chi, Taiwan[38,39*]	2420	1250			
Watershed PL-11		1100	17.8	308	-290
Watershed PL-12		1400	17.6	408	-390
(24) Chiengmai, Thailand[40*]	2035	930	16.0	4.0	+12.0
Ibidem applying 1969 nutrient concentrations in rainfall[41]			5.1	4.0	+1.1
(25) San Eusebio, Venezuela[42+]	1500	565	5.6	1.7[D]	+3.9

*catchment based study
[a-m]see footnotes next pages
+(zero-)tension lysimetry
°total soluble amounts unless indicated otherwise
▲excluding fixation inputs and denitrification losses

Magnesium			Potassium			Phosphorus			Nitrogen		
I	L	I-L	I	L	I-L	I	L°	I-L	I	L	I-L▲
(kg ha^{-1} yr^{-1})											
4.0	30.5	-26.5	9.6	22.0	-12.4	1.2[n]	0.7[n]	+0.5[n]	15.4[p]	10.6[p]	+4.8[p]
	43.1	-39.1		49.0	-39.4		5.9	-4.7		38	-23
2.8	8.5	-5.7	2.0	3.6	1.6	0.08[s]	0.03[s]	-0.01[s]	1.1[t]	19.4[t]	-18.3[+]
2.6	5.0	-2.4	5.4	21.6	-16.2	0.17[s]	0.03[s]	+0.14[s]	1.7[t]	-5.6[t]	-3.9[t]
1.4	90	-89	3.4	76	-72						
	78	-77[v]	13.5[v]		-10[v]						
0.3	51	-50.7	0.8	14.9	-14.1						
3.4[x?]	45	-42	3.7	20	-16						
	17[v]	-13		3.4[v]	+0.3						
1.2	10.3	-9.1	4.6	12.1	-7.5	0.2[s]	tr.[s]	+0.2[s]	7.7[t]	7.4[t]	+0.3[t]
5	44	-39	9.5	9.3	+0.2	1	0.7	+0.3			
22	72	-50	21	18	+3		0.7[y]		1[z]	2.3[z]	-1.3[z]
1.3	(25)	(-24)	7.3	(22)	(-15)	0.53			6.5		
4.9	15	-10.1	18.2	20.8	-2.6	0.9	1.3	-0.4	14[B]	29[B]	-15[B]
3.9	166	-162	7.3	10.0	-2.7		tr.		9.7[C]	1.6[C]	+8.1[C]
3.9	220	-216	7.2	10.4	-3.2		tr.		9.6[C]	1.6[C]	+8.0[C]
16.1	2.5	+13.6	12.3	2.6	+9.7						
0.2	2.5	-2.3	0.6	2.6	-2.0						
5.2	0.6[D]	+4.6	2.6	2.2[D]	+0.4	1.1[E]	0.3[D]	+0.8	9.9	4.4[D]	+5.5

Table 3 continued (footnotes)

[18]Abdul Rahim & Zulkifli (1986); [19]Zulkifli (1989); [20]Zulkifli et al. (1989)
Undisturbed Dipterocarp forest; deeply weathered granite; deep ultisols (2/3 rd) and oxisols (1/3 rd) of sandy clay (loam) texture; catchment area 29.6 ha; observation period for streamflow quantity and quality (July 1980 - June 1983) included a wet and a dry year with overall precipitation total close to long-term average; rainfall chemistry studied between September 1986 and August 1987 (normal rainfall); possibility of sample enrichment for calcium and magnesium (dust) cannot be ruled out; [k]phosphate phosphorus; [l]inorganic forms only, total nitrogen input 11.4 kg $ha^{-}yr^{-1}$; [m]applying a runoff figure of 575 mm yr^{-1} to account for deep leakage and using concentrations of calcium, magnesium and potassium in rainfall as determined at nearby Pasoh (Manokaran 1980).

[21]Bruijnzeel (1983a); [22]Bruijnzeel (1983c); [23]Bruijnzeel (1984)
Plantation forest of _Agathis dammara_ (11-35 years old); andesitic tuffs underlain by andesitic breccias; rather fertile andept; catchment area 18.7 ha; observation period December 1976 - February 1978, including severe dry spell; weekly sampling; outflow data normalized for average runoff total; [n]phosphate-phosphorus and based on limited data; [p]based on limited number of samples taken in wet season; [q]values corrected for incorporation in aggrading biomass (see Bruijnzeel (1983a) for details) not necessarily representative of outputs for ecosystem in steady state.

[24]Parker (1985)
"Tropical wet" forest; Quaternary basaltic volcanic deposits; "ultic" Andept; drainage rates approximated by site water balance technique using Penman-Monteith evaporation model; [r]observation period 404 days (March 1983 - May 1984) with below-average rainfall; quoted drainage rate would correspond with unrealistically high value for forest ET (2190 mm yr^{-1}); concentration of percolating water obtained from "numerous" suction lysimeters at 70 cm with suction maintained at -300 mbar and sampled every two weeks; rainfall often sampled on an event basis; high standards for collection and analysis of samples; [s]phosphate-phosphorus; [t]inorganic forms; [u]average annual nutrient input as given by Parker (1985); output based on streamwater chemistry by Parker (1985) combined with runoff estimated as rainfall minus average lowland rainforest ET of 1425 mm yr^{-1} (Bruijnzeel 1989b).

[25]Crowther (1987a); [26]Crowther (1987b)
Lowland rainforest; limestone; 40-70 cm deep Rendolls (?) on footslopes with alluvial admixture; amounts of drainage estimated by subtracting forest ET (approximated as 0.83% of "class A" pan evaporation at the nearest meteorological station) from rainfall; nutrient concentrations in drainage water obtained by 3-weekly sampling of one zero-tension lysimeter inserted at contact with bedrock and also [v]by 3 to 6-weekly sampling of large numbers of seepage points in caves for one year; drainage rates considered slight underestimates by the investigator himself; incident precipitation sampled "regularly"; no further details given on collecting procedures.

[27]Turvey (1974)
Colline rainforest, basaltic volcanic agglomerates underlain by Mg-bearing phyllites; "acid red to brown clay soils"; catchment area 16.25 km^2; observation period July 1972 until May 1973 with numerous gaps in records; rainfall (after December 1972) and streamflow sampled at least weekly; [w]nutrient concentrations in rainfall low and at least for calcium considered anomalous (analytical error) by investigator himself.

[25,26]Crowther (1987a,b)
Environmental and procedural details as given for Gua Anak Takun study (17); drainage of soil water sampled by two lysimeters; [x]enriched by dust from quarry.

[28]Horng et al. (1985)
Warm temperate rain forest (evergreen); sandstone and shales; inceptisols of fine silt loam texture; catchment area 3.4 ha; observation period 1984; rainfall sampled per event, streamflow at least daily; catchment probably leaky as nearby catchments exhibited consistently higher runoff totals (ca. 300 mm yr^{-1}).

[29]Golley et al. (1975)
"Evergreen seasonal" forest; shale interbedded with dolomite and calcareous sandstone; black clay soils (Vertisol) with high base saturation; an approximate nutrient budget was computed for the 259 km^2 Sabana catchment over 1967: rainfall data from various stations in the region were combined to give annual total of 1933 mm; rainfall at one location sampled on seven occasions in September 1967, possibly by using a plastic sheet; discharge data available between May and December and extended to annual value by comparison with other catchment; chemical composition of river water is average of a number of streams in the region and based on a very limited number of samples; concentrations for Rio Sabana itself far higher than regional average.

[30]Jordan et al. (1972); [31]Clements & Colon (1975); [32]Brown et al. (1983); [33]Lugo (1986) Lower Montane Rainforest (500-1000 m) grading into dwarf forest at highest elevations; andesitic rocks; generally acid ultisols of clay loam texture and rather low base saturation; catchment area not specified but estimated at ca. 10 km^2 from general maps in [32]; approximate nutrient budget computed by present writer using average rainfall and streamflow data from [33]; streamwater quality data from USGS quoted in [33] (numerous samples over several years); precipitation chemistry: average concentrations for El Verde at 500 m from [30] and [31] (two separate full years) and for dwarf forest at 1000 m (Trinidad Pizarro (1985) and personal communication in [33]) weighted by elevation zones as given in [33]; upper 10% of catchment received extra input of 10% by "cloudstripping" (Weaver 1972) with chemical composition as determined by Trinidad Pizarro in [33]; [y]phosphate-phosphorus; [z]nitrate-nitrogen, input as determined at top end of watershed; input of ammoniacal nitrogen of similar magnitude ([33]); if concentrations of total nitrogen in rainfall as reported for El Verde at 500 m (1970) are extrapolated to entire watershed an input of ca. 23 kg ha^{-1} yr^{-1} is obtained.

[34]Aitken et al. (1972); [35]Edwards (1982)
Lower Montane Rainforest at 2450 m; andesitic volcanic tuffs; inceptisols; amount of streamflow estimated by subtracting forest ET (as determined from regional ET-elevation-rainfall relationship[34]) from rainfall; composition of streamflow based on single baseflow sample taken in July 1971; baseflow composition at that time of year probably within 10% of annual mean (Bruijnzeel 1983a); nutrient input based on bi-weekly readings of rainfall amounts in two clearings and 9 samples each between December 1970 and August 1971; 1 cm wire mesh in funnels, no preservatives.

Table 3 continued

[36]Jordan (1969); [37]Edmisten (1970); [32]Brown et al. (1983)
Lower Montane Rainforest at 500 m; andesitic rocks; acid topsoil overlying yellow clay and red/yellow clay/rotten rock; rainfall (2 collectors above canopy) and streamflow (catchment area unknown) sampled weekly between October 1967 and September 1968; [A]as given in [32], Jordan (1969) may have used different values (not specified); [B]based on limited data collected in December-February by [37].

[38]King & Yang (1984); [39]Cheng et al. (1987)
Warm-temperate coniferous forest on slopes and warm-temperate rainforest in valleys; sites at 2550 m and at 24° N.L.; slates; brown podzolic soils with silty to sandy loam texture; catchment areas 144 ha (PL-11) and 238 ha (PL-12); vegetation in PL-11 denser; rainfall and streamflow monitored since 1969; sampling for chemical analysis between January 1981 and December 1982 (anions 1982 only) with precipitation ca. 8% above longterm means; [C]inorganic forms of the element only; total outputs of soluble nitrogen ca. 2.1 kg ha^{-1} yr^{-1} for both catchments.

[40]Naprakob et al. (1976); [41]Watnaprateep (1984)
Evergreen hill forest dominated by *Castanopsis*; granodiorite?; "latosol" of sandy clay loam texture, probably very poor; catchment area 65 ha; rainfall and runoff monitored since 1965; weekly sampling of rainfall and streamflow in 1974-1975; no further experimental details given; concentrations in rainfall rather high, the ones quoted for 1969 are lower and similar to those of Manokaran (1980).

[42]Steinhardt (1979)
Lower Montane Rainforest ("cloud" forest); silty and clayey sedimentary rocks; humitropepts with variable drainage; drainage rate approximated by subtracting forest ET (as determined by energy balance method) from rainfall; chemical composition of drainage water determined with three suction lysimeter plates with suction set at levels in surrounding soil (twice/week); [D]nutrient outputs considered underestimate by investigator himself; rainfall sampled from plastic gauges at least weekly between December 1983 and November 1984 (rainfall 10% higher than normal); [E]samples considered contaminated with insect debris by investigator.

from various publications. Although perhaps a rather bold thing to do, the exercise was considered justified in view of the limited amount of published information available for the montane tropical environment and the rather flimsy hydrological foundations on which some often quoted budgets (e.g. no's 1, 3, 7, 16 and 19 in Table 3) are based. Details on measuring and computation procedures for all studies quoted in Table 3 are given in footnotes.

Sites were subdivided as follows: first, a distinction was made between lowland forests (n=19) and montane forests (n=6). The lowland sites were then grouped according to the nature of the substrate, i.e. soils with moderate to very low fertility (n=12), and soils with moderate to high fertility (n=7). Most of the forests belonging to the low fertility group are on Ultisols and Oxisols, with one site on Spodosols.

The variation in reported nutrient fluxes, not only per element and fertility group but also per lithology and even for particular study sites (sic!), is considerable (Table 3).

For example, reported calcium and magnesium inputs for the Oxi/Ultisol group range from barely detectable at sites no. 10 and 11 in the Central Amazon to 42 and 17 kg ha^{-1} yr^{-1} respectively at site no. 12 (Peninsular Malaysia). Similarly, losses for these elements vary between negligible in the Amazon and 47 and 30 kg^{-1} ha^{-1} yr^{-1} respectively at site 6 (Ivory Coast). Although such differences might theoretically reflect differences in rainfall amounts and regimes, and/or bedrock type and degree of weathering in contrasting environments, this becomes less probable when finding strikingly different net losses or gains for catchments experiencing comparable climatic and geological conditions (e.g. sites 4, 7 and 12, which are all on deeply weathered granites in the Far East, or sites 3 and 10/11, all in the Amazon and on very infertile substrates).

Rather than taking such findings at face value, one would be well advised to investigate the possibilities of contamination/enrichment of precipitation samples by regional dust (Roose 1981; Crowther 1987a; Yusop et al. 1989), fire (Edwards 1982; Jordan 1982) or organic debris (Steinhardt 1979). Also, difficulties with the analysis of highly dilute water samples (Turvey 1974; Jordan 1982; Poels 1987) and the underestimation of streamflow amounts due to basin leakage (all Amazonian sites with sandy valley fills (Bruijnzeel 1989b; cf. Abdul Rahim & Kasran 1986) or the use of too high an estimate for forest ET (producing an underestimate of the mount of drainage; Jordan 1982; Parker 1985) sometimes influence the results.

It is perhaps significant in this respect that Bruijnzeel (1989a) concluded from a comparison of average elemental concentrations of bulk precipitation at a number of stations in the humid tropics, that the results often seemed to reflect the rigidity of sampling procedures (including positioning of collectors) rather than environmental factors like proximity to an ocean, volcanism or climatic seasonality (see also Lewis et al. 1987). It is quite possible, therefore, that many studies of elemental fluxes in tropical forest areas have overestimated atmospheric nutrient inputs, although the effect will be moderated to an unknown extent by the fact that contributions of dry deposition on forest canopies are largely unaccounted for by traditional collecting devices. Further work on this aspect is as desirable as difficult (White & Turner 1970). Some progress however has been made in this respect in relation to such problems as acid deposition and forest dieback in the temperate zone (e.g. Prupacher et al. 1983; Gosz et al. 1983; Lovett & Lindberg 1984; Lindberg et al. 1986; Delleur 1989; cf. Lewis et al. 1987).

Detecting procedural deficiencies is one thing, correcting them quite another. However, in some cases better hydrological information and alternative rainfall chemistry data for the same or a nearby location was used to obtain more realistic results. For instance, the use of an alternative data set for site 3 (San Carlos, Venezuela) transformed this apparently rather vigorously nutrient accumulating forest (Jordan 1982) into a system with marginal losses, which is much more in line with results obtained for similar forest on very poor soils elsewhere in the Amazon (Brinkmann 1983; Franken & Leopoldo 1984; sites 10 and 11; see footnotes in Table 3).

Repeating the procedure for another apparently nutrient accumulating system (site 12, Bukit Berembun, Malaysia) yielded a similar improvement in that the corrected net solute outputs were much closer to those reported for granitic terrain in Hong Kong (site 4; Lam 1978) than the previous ones (Table 3).

As such, it remains to be seen whether the Surinam forest (site 9 in Table 1; Poels 1987) is indeed accumulating calcium and potassium to the extent claimed by the investigator, especially when taking into account the difficulties that were met in collecting and analyzing representative precipitation samples as well as the possibility of unrecorded deep flow through the sandy valley fills (cf. Table 1).

3.4.2 Lysimetry- versus catchment based estimates

The discrepancies between estimates of hydrologic nutrient outputs based on soil- or groundwater sampling on the one hand and streamflow sampling on the other, become readily clear from the results obtained at several locations where both techniques were used (sites 1, 6, 8, 15 and 17, to a lesser extent also site 14).

From the data reported for the Spodosol site at San Carlos (site 1; Herrera 1979), one could either conclude that the forest is losing nutrients (zero-tension lysimetry) or is accumulating them (samples taken from a small stream). An alternative computation by the present writer also suggested slight accumulation (see footnotes in Table 3). Conversely, an intensive study of precipitation and streamflow chemistry for the much larger (47500 km^2) Caura river basin, a "blackwater" area some 500 km northeast of San Carlos draining equally infertile soils, indicated high gross and net losses of calcium, magnesium and potassium. Phosphorus was retained (site 2; Lewis 1986; Lewis et al. 1987).

Similarly, large losses of calcium, magnesium and potassium have been reported for an old secondary forest in the Ivory Coast (site 6) on the basis of zero-tension lysimetry by Roose (1981), whereas the same forest would be accumulating these elements on the basis of (headwater) stream sampling. The opposite pattern was reported for phosphorus. Roose's findings were confirmed for calcium, but not for magnesium and potassium, by Bernhard-Reversat (1975) for a similar forest nearby (site 8).

How are such apparently conflicting results to be explained? In the case of nutrient poor systems, where the surface rootmat has been shown to be an efficient filter for incoming nutrients (Stark & Jordan 1978; Brinkmann 1983), one may indeed expect a slight decrease in soil water solute levels with depth as further nutrients are taken up by roots in the subsoil without being replaced by actively weathering minerals. As such, somewhat larger nutrient losses may be obtained on the basis of (zero-)

tension lysimeters installed at relatively shallow depths in these depleted soils than on the basis of streamwater chemistry (cf. the results obtained by Russell (1983) and Poels (1987), sites 5 and 9 respectively), provided the stream has not reached the *active weathering front* (i.e. the contact between weathering mantle and fresh bedrock (Bruijnzeel 1989b).

As shown by the Caura basin study (cf. site 20 in Table 3), streams draining areas with infertile soils may still carry considerable amounts of nutrients *once the stream has exposed the fresh bedrock*. The point is neatly illustrated by the chemical analyses for a deep Oxisol on granite in Peninsular Malaysia presented by Burnham (1989). Soil cationic concentrations were very low throughout the profile (which included a substantial portion of weathered rock) and rose sharply in a narrow band close to the fresh rock at a depth of ca. 10 m. Since streams draining this type of terrain carry fair amounts of cations (Douglas 1967b; Kenworthy 1971; Abdul Rahim & Yusop 1986), it cannot be excluded that the latter derive to a large extent from this narrow zone of active weathering (Bruijnzeel 1989b).

Nutrients released by rock weathering at such depths are essentially out of reach of the root network (Baillie 1989). Therefore, although clearly a loss to the ecosystem at large, these nutrients do not play a role in the nutrient economy of the forest itself. It follows that in such cases the dissolved nutrient load of a stream is not representative of the nutrient loss experienced by the vegetation. Experimental work is needed to test the above hypothesis and may usefully combine observations of root distribution (Baillie & Mamit 1983) with determinations of soil- and soil water composition throughout the weathering profile at different topographic positions (Verstraten 1980; Bruijnzeel 1983a).

Discrepancies in nutrient exports obtained via lysimetry or streamflow sampling should be relatively small in the case of deep and more fertile soils, where contrasts in nutrient contents of bedrock and soil material are less pronounced and competition for nutrients in the topsoil is less severe. Nevertheless, water emerging as springs in such areas is usually somewhat more concentrated than soil water sampled higher up in the profile (Bruijnzeel 1983a; Parker 1985), reflecting differences in residence times. Interestingly, ecosystems on substrates in which only a certain nutrient is particularly scarce, e.g. potassium in limestones or ultrabasic rocks, exhibit an oligotrophic cycling strategy (Jordan & Herrera 1981) with respect to that nutrient, but not for other elements. For example, Crowther (1987b) found much higher potassium concentrations in soil water intercepted by zero-tension lysimeters at the (rather shallow) soil-bedrock (limestone) contact in West Malaysia than in deep groundwater seeping into the underlying caves (sites 15 and 17 in Table 3). Apparently, deep roots in fissures and joints were able to extract considerable amounts of potassium from the percolate. Discrepancies between the two types of samples were generally much less for calcium and magnesium, which were abundantly available. Similarly, potassium concentrations in the soil solution (as sampled by suction lysimetry) decreased rapidly with depth in montane rain forests on ultrabasic rocks in Sabah, with very low amounts present in stream water (L.A. Bruijnzeel & M.J. Waterloo, unpublished data). Although seepage water was in contact with the fresh rock in both the above examples, no rise in potassium concentrations was recorded, reflecting the inability of the rock to supply this element in significant quantities.

The examples given sofar do not only confirm the methodological limitations referred to in the previous section, they also provide some guidance as to which method (the lysimetric or the catchment approach) is the more representative when it comes to evaluating nutrient losses from forest ecosystems. It would seem as though carefully selected small, yet watertight, catchments, supplemented by hillslope plots if the spatial variation of soil and vegetation requires so, and monitored for a number of years to account for climatic variability, could give the best estimates of ecosystem nutrient losses (Clayton 1979). Estimates based on various forms of lysimetry alone should be treated with caution in view of the above-mentioned problems related to the estimation of drainage and average soil water composition.

Highly infertile soils present a special case in that different approaches are needed, depending whether one is primarily interested in the nutrient loss from the biologically active portion of the ecosystem or from the system at large. In the former case, either the lysimetric or the small catchment approach may do, in the latter a sufficiently large catchment area is needed.

3.4.3 Patterns of calcium, magnesium and potassium losses

During the discussion presented thus far one may well have wondered to what extent it is possible to derive any meaningful generalizations from the varied data in Table 3. However, scatter plots of annual calcium, magnesium, and potassium outputs against corresponding amounts of streamflow for most (small) catchment studies listed in the Table show that this is indeed feasible to a certain extent (Figures 12a-c). Despite some overlap one can clearly distinguish four groups with contrasting nutrient export patterns. Figure 12d presents the range of values associated with the respective groups.

The sites of the *Spodosol/highly depleted Oxisol group* (mainly Amazonian studies, also site 24) consistently exhibited very low losses of calcium and magnesium, and often of potassium as well. The highest values in this group were reported for "extremely poor" (Buschbacher 1984) Ultisols near San Carlos de Rio Negro (alternative computation, Table 3). Since these estimates are based on zero-tension lysimetry (Buschbacher, unpublished data, quoted by Medina & Cuevas 1989), they may represent slight overestimates, although it may well be argued that they reflect a somewhat less weathered substrate, the soil after all having been classified as an Ultisol rather than an Oxisol. As such, this site may represent a transition to the second group (see below).

Nutrient losses for the *Oxisol/Ultisol group* (n=5) are somewhat larger and more variable than those observed for the most infertile sites, although there is some overlap for potassium (Figure 12d). Nevertheless, the present group average for potassium (about 11 kg ha^{-1} yr^{-1}) is much higher than that for the first group (< 2.7 kg ha^{-1} yr^{-1}).

Still larger losses of calcium and magnesium have been recorded for sites that are largely on *Inceptisols* (n=5). Although again there is an overlap with the preceding group for potassium, average values for the two groups are quite different at 23 versus 11 kg ha^{-1} yr^{-1} respectively.

Finally, very high losses of calcium have been reported for forests on calcium-bearing rocks with *Mollisols/Vertisols* (sites 15, 17, and 19 in Table 3). The corresponding outputs of magnesium (variable) and potassium (low and similar to the Oxisol/Ultisol group) reflect the nature of the underlying rocks (cf. sites 16 and 23, which are both on magnesium bearing rocks and hence exhibit fairly high losses for this element).

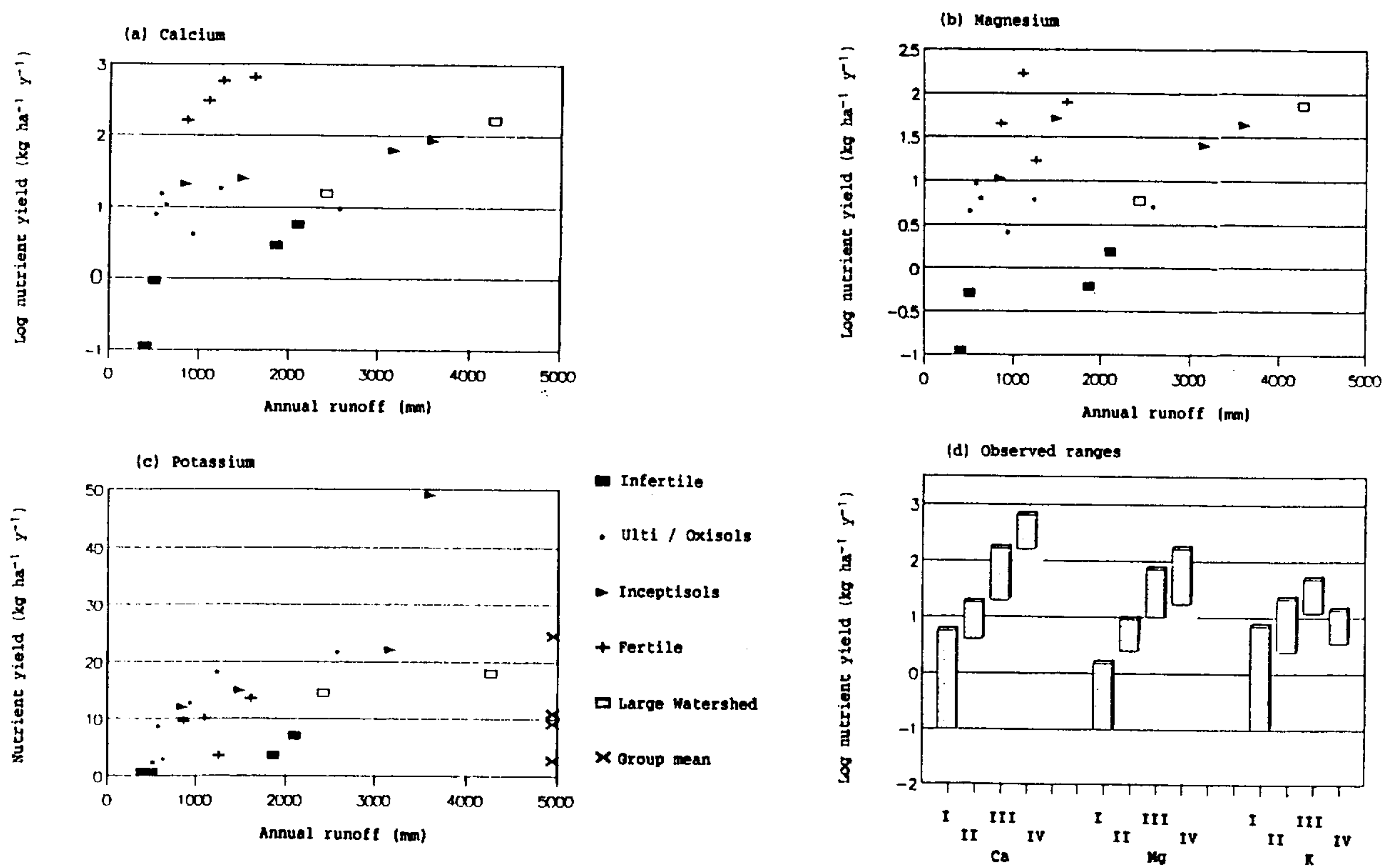

Figure 12. Scatter plots of annual runoff versus nutrient yield for 20 selected (sub)tropical forest ecosystems (a) calcium, (b) magnesium, (c) potassium, (d) observed ranges.

Patterns of calcium, magnesium or potassium losses from *Montane forests* do not seem to deviate from those for lowland forests (Table 3 and Figure 12; see also the next section).

3.4.4 Phosphorus and nitrogen

Phosphorus appears to be accumulating in almost all studies quoted in Table 3, reflecting the very low mobility of the element. It has been suggested that most of the phosphorus released by weathering becomes tied up in organic and ferro-aluminium compounds in the soil (Sanchez 1976; Clayton 1979). As such, one cannot expect the high amounts of phosphorus reportedly released by the weathering of the volcanic tuffs at site no. 13 (and largely incorporated in the rapidly growing vegetation; Bruijnzeel 1983a,b) to show up in streamflow after removal of the vegetation (cf. Chapter 5). Incidently, this low mobility of phosphorus, apart from the various possibilities mentioned earlier, also renders the extremely high phosphorus exports reported for San Carlos (no's 1 and 3 in Table 3) highly unlikely.

Nitrogen also represents a special case in that its biogeochemical cycle includes a number of processes that involve the element in its gaseous form (Odum 1971). A significant portion of nitrogen inputs to the forest ecosystem occurs via biological fixation. The representativity of the data on (atmospheric) inputs and (hydrologic) outputs of nitrogen presented in Table 3 is limited still further by the variation in nitrogen components covered by the various studies as indicated in the respective footnotes. Conclusions of net losses or gains based on inorganic nitrogen constituents alone are easily overturned by the inclusion of the organic fraction (Brinkmann 1983). Therefore, the remark of Vitousek & Sanford (1986) with respect to the possibility of detecting any patterns for tropical forests on the basis of available data is especially valid for this element and no further attempt at analysis has been made here.

More or less comprehensive estimates of the various inputs and losses of nitrogen have been made for only two tropical forests, the Oxisol site at San Carlos (site 3 in Table 3; Jordan et al. 1982) and the lower montane rain forest at El Verde, Puerto Rico (site 22; Edmisten 1970). Although limited in number, the results neatly illustrate the limitations of the small watershed technique in this respect.

For example, only 42 per cent of the total nitrogen input at San Carlos entered the forest via bulk precipitation, whilst 58 per cent was supplied by biological fixation. Also, hydrologic losses made up 83 per cent of the total loss, with denitrification accounting for the remaining 17 per cent (Jordan et al. 1982). In other words, the (inorganic) nitrogen budget at this site was transformed from a net loss of 2.6 kg ha^{-1} yr^{-1} (Table 3; hydrologic inputs and outputs only) to a net gain of 8.6 kg ha^{-1} yr^{-1} when all processes were taken into account.

It is difficult to envisage, however, how a forest which is presumably in a state of dynamic equilibrium (Uhl 1982), would be accumulating this amount of nitrogen. Alternately, this result may reflect the fact that concentrations of organic nitrogen were not determined. Elsewhere in the Amazon ("clear water" areas near Manaus), organic nitrogen constituted 78 and 88 per cent of the total hydrologic gains and losses of nitrogen respectively (Brinkmann 1983).

In contrast to the observations at San Carlos, those at El Verde were made during a rather short period with relatively low rainfall totals (Edmisten 1970). Fixation on leaf surfaces and by root nodules accounted for 86 per cent of the total nitrogen input, i.e. only 14 per cent was supplied via precipitation. Also, losses in drainage water (29 kg ha^{-1} yr^{-1}) were much smaller than those associated with denitrification (ca. 56 kg ha^{-1} yr^{-1} or 66 per cent of the total loss). Again, an apparent net loss of 15 kg ha^{-1} yr^{-1} (hydrological budget; Table 3) was turned into a net gain of 17 kg ha^{-1} yr^{-1} after including biological transfers of nitrogen.

Montane rain forests, like "heath forests", are generally considered to be short of nitrogen, although this contention is based on analyses of individual processes, such as litterfall and mineralization rates in the soil (Grubb 1977; Vitousek & Sanford 1986; Marrs et al. 1988) rather than on a quantification of all processes involved (Edmisten 1970). In view of the apparently conflicting evidence, further work, preferably of an integrated nature, is desirable.

Summarising, nutrient losses from tropical forest terrain are primarily governed by the nature of the geological substrate and the degree to which this has been weathered. A distinction has to be made between losses from the ecosystem at large and losses from the biologically active portion of the ecosystem, especially for oligotrophic sites.

The present chapter concludes our description of the undisturbed tropical forest environment. The next two chapters will discuss the hydrological and chemical responses of the ecosystem to various kinds of disturbance by man.

4 HYDROLOGICAL IMPACTS OF (DE)FORESTATION

4.1 Introduction

Hunter-gatherer societies have lived in tropical rain forest regions for thousands of years without destroying the forest (Lea 1975). Although some parts of the forest were cleared long ago for permanent agriculture (e.g. the alluvial plains of Asia, which formerly carried swamp forest, for the cultivation of irrigated rice), for a long time the most extensive farming in rain forests all over the tropics has been by "shifting cultivation", also known as "slash and burn agriculture" (Whitmore 1990).

Shifting cultivation has been identified as the single most important reason for loss of tropical forest before the 1980's (Lanly 1982). It was estimated to be responsible for about 35 percent of all deforestation in Latin America, whilst the corresponding figures for Africa and Asia read 70 and 50 percent respectively (Hadley & Lanly 1983). Jackson (1983) estimated that about 150 million people are involved in traditional agriculture in the tropical forest zone.

Basically, shifting cultivation consists of (manually) felling a patch of forest, usually at the start of the period of least rainfall, allowing the material to dry, burning the slash shortly before the rainy season, and planting rapidly maturing crops in the ash (Plate 3). After one or two crops have been harvested, yields diminish and the original field is abandoned to forest regrowth (the so-called "bush fallow"). The farmers then move on to another piece of forest and the cycle is repeated. Since it is easier to fell and burn secondary forest than virgin jungle, repeated rotation through an area is often preferred by shifting cultivators to continual movement into new areas (Watters 1971; Scott 1987). A number of variations on this basic theme can be distinguished (Chuasuwan 1985).

In contrast to the view prevailing a few decades ago (FAO 1957), it is now more or less generally accepted that shifting cultivation is a form of agriculture that is sustainable under the general climatic and edaphic conditions of the humid tropics, as long as it is practiced within the limits of the ecosystem's capacity to regenerate. However, when either the cropping period is extended too long (impairing forest succession: Uhl 1987), or the bush fallow period becomes too short (limiting the build-up of nutrients in the vegetation to be released upon burning), the system will degrade (Zinke et al. 1970; Sanchez 1976; Hatch 1983; Scott 1987; see also Chapter 5).

Although rain forests are capable of supplying a host of minor products (Lea 1975; Myers 1988a) and have done so for a long time, there has been a rapid change in the way these forests are valuated by modern man. Nowadays, rain forests are mainly regarded as a source of timber. For example, just before the second world war, the relative importance of timber and minor forest products traded from Indonesia was 55 to 45 per cent. Today, it is 95 to 5 per cent (Whitmore 1990).

Estimates of the rate at which moist tropical forests were being altered by man in the early eighties lie around 12 million ha yr^{-1}, 63 per cent of which (7.5 million ha) was cleared, with the remainder (4.4 million ha) "selectively" harvested. Another 4 million ha yr^{-1} were reportedly cleared in the more seasonal tropics (Lanly 1982). These figures are probably conservative to the extent that national reports to FAO may well have been over optimistic. In addition, the great forest fires occurring in Borneo in August - October 1982 and in March - May 1983 (damaging more than 4 million ha of (partially logged-over) forest; Malingreau et al. 1985) and the deliberate burning of 16 million ha (!) of forest along the southern fringe of Amazonia in 1987 and 1988 for the creation of pastures (Whitmore 1990) are not included in the above overall estimates.

The causes and patterns of forest clearance vary considerably between regions and a detailed analysis of these is beyond the scope of the present report (see overviews by Myers 1980; Lanly 1982; Whitmore 1990). Now that satellite imagery is becoming more widely available, chances for adequate monitoring of rates of forest removal are improving (Myers 1988b). Recent reports on local or regional situations include those by Gentry & Vasquez (1988) (Peruvian Amazon), Fearnside (1987) (Brazilian Amazon), Malingreau & Tucker (1988) (southeastern Brazil), Eyre (1987) (Jamaica), Sader & Joyce (1988) (Costa Rica), Pullan (1988) (West Africa), Hirsch (1987) (Thailand), Quinnell & Balmford (1988) (Philippines) and Smiet (1989) (Indonesia).

As indicated earlier, the general term "deforestation" is rather meaningless as a descriptor of land-use change and each case needs to be defined properly (Bruijnzeel 1986; Hamilton 1987). In this and the next chapter, three levels of intensity of forest disturbance, viz. low, intermediate and high (Jordan 1985), will be distinguished when discussing the environmental impacts of "deforestation".

Low-intensity types of disturbance (mainly dealt with in Chapter 5) include such small-scale and short-lived events as natural tree falls and small clearings. Apart from the already mentioned forest fires and slash and burn agriculture, both of which generally produce a temporary effect, selective logging of forests may also be ranked as a disturbance of (at least) intermediate intensity, depending on the volume of timber removed and the type of equipment used (Horne & Gwalter 1982; Plates 4-8).

The partial removal of timber from a forest stand may be interpreted as the creation of a large number of variably sized gaps (Jordan et al. 1985). Since modern logging techniques are generally highly mechanised, such an operation implies the creation of access roads, skid (or "snig") tracks and landings, and in doing so a considerable portion of the area that is to be logged will become disturbed (Burgess 1971, 1975). Often, the lay-out, construction and maintenance of logging roads, etc. is poor and former skid tracks and landings may remain compacted for many years after their creation (Malmer & Grip 1990; Van der Plas 1990) and will influence runoff and regeneration patterns accordingly (Plates 6 and 7).

As such, selective logging will usually have to be classified as a disturbance of moderate intensity at least, also because felling and extraction of large trees may produce so much damage to the surrounding vegetation that regrowth may be too slow for further profitable exploitation (Burgess 1971; De Graaf 1986). Finally, roads make logged-over forests more accessible to settlers, hunters, etc., thereby increasing the risk of further degradation (Wyatt-Smith 1987).

Generally, a forest subjected to one of the above-mentioned types of disturbance may recover to its previous state if left alone for a sufficiently long period (Saldarriaga 1987; Riswan & Kartawinata 1988). Clearly, this is not the case when forest is converted to permanent agriculture (grazing, cropping, extractive tree crops) or production forestry and these must all be classified as disturbances of high intensity.

Forest land can be cleared in a number of different ways, each characterised by a certain degree of soil disturbance. Couper et al. (1981)

compared the effects of several clearing methods, ranging from traditional slash and burn through modernised manual clearing (using chainsaws) to highly mechanised techniques, in terms of man hours and energy expenditure, whereas Lal (1981) reported on surface erosion rates associated with the various techniques as observed at the same site. Although manual clearing was slowest and the most expensive of the methods investigated, soil erosion in the first year after clearance amounted to only 0.4 t ha^{-1} (Lal 1981). Conversely, clearance by means of crawler tractor with a shear blade attached, although the most rapid and economical, induced an erosion rate of almost 4 t ha^{-1} yr^{-1}. Finally, crawler tractors with tree pusher/root rake attachments were more expensive to use than the ones equipped with a shear blade only and produced an erosion rate of more than 15 t ha^{-1} yr^{-1}. All values cited were for no-tillage agriculture during the first year after clearing (Couper et al. 1981; Lal 1981).

Similarly, Van der Weert (1974) and Seubert et al. (1977) drew attention to the negative effects of mechanised clearing on root development and therefore on agricultural production, mainly through increased soil compaction. More recently, the experiments of Dias & Nortcliff (1985a) revealed a close correlation between the number of tractor passes over an oxisol in Amazonia and the resulting degree of soil compaction. Uhl et al. (1982, 1988b) reported slow regeneration of natural regeneration following clearing by bulldozer in areas underlain by Podzols and Oxisols in Latin America.

Although there can be no doubt that manual methods of forest clearing are far superior to most modern methods of clearing in terms of damage done to the forest floor (Dias & Nortcliff 1985b), there will often be no choice but to resort to mechanical means. However, as pointed out by Couper et al. (1981), bringing new land under cultivation should not just be done with the objective of carrying out as much as possible as "cheaply" as possible, since this approach could well result in irreversible damage to the land. The short-term gains would then be more than offset by the costs needed to restore or maintain soil productivity in the long term (Van der Weert 1974; Seubert et al. 1977; Ollagnier et al. 1978).

Martin (1970) and Van der Weert (1974) have given a number of simple suggestions as to how soil damage during clearing could be minimised. These include the proper choice of the timing of the operation (preferably during times of low soil moisture levels as wet soil is easier to compact), the avoiding of the use of tree-pushing/root raking equipment wherever possible (i.e. leave stumps to rot), and the minimising of the number of tractor passes during windrowing by optimising the distance between windrows and burning the slash before windrowing. By planting crops close to the windrows the reduction in windrow spacing should not affect accessibility in the newly established plantation too seriously.

Couper et al. (1981) stressed the importance of employing skilled tractor operators since bulldozers were originally designed to move soil rather than clear forest. In addition, the regular sharpening of tractor shear blades greatly increased the efficiency of the method. Provided all of the above cautionary measures are taken, Couper et al. (1981) reckoned shear blade clearing to be capable of providing a suitable base for sustained yields in the future.

Although sky-line logging (also called "high-lead yarding") has been shown to be capable of minimum disturbance of the surface in steep terrain (Pearce & Griffith 1980; Plate 8), the degree of damage done to the vegetation surrounding the harvested trees makes this technique more suitable for use in clearcutting than in selective logging operations.

In the following, the effects of partial and complete forest removal c.q. planting on climate (notably rainfall; section 4.2), water yield (section 4.3) and its seasonal distribution (section 4.4) as well as on sediment production (section 4.5) will be looked at in detail.

4.2 Effect on rainfall

Differences in micro-climatic conditions near the ground in tropical rain forests and large clearings are well-documented (see Richards (1952) and Schulz (1960) for excellent discussions of the early literature; also Pinker 1980; Lawson et al. 1981; Ghuman & Lal 1987).

In general, near-surface conditions in clearings are much more harsh than inside the forest, with greater insolation, higher maximum temperatures and vapour pressure deficits and, especially, much higher soil temperatures, resulting in a strongly increased evaporative demand of the atmosphere (Figure 13). The direct ecological (e.g. decomposition of organic matter, microbial and soil faunal activity, etc.) and site-hydro-

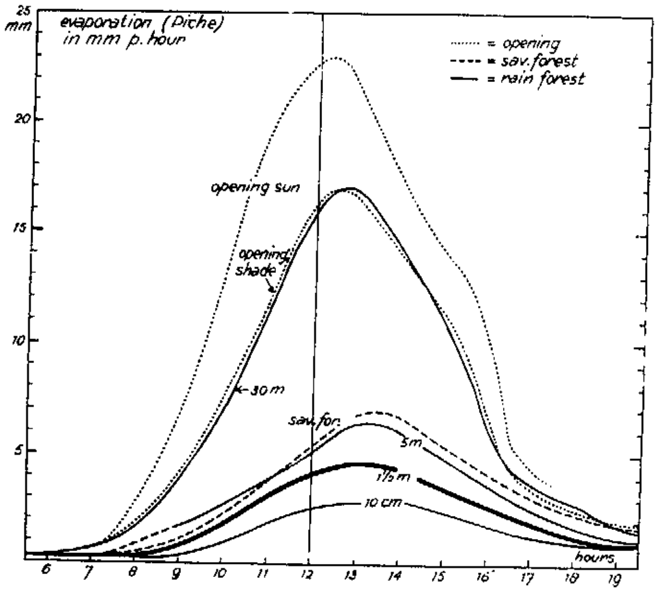

Figure 13. Daily march of Piche evaporation at several heights in rain forest, savana forest and a large clearing during the dry season of 1957 in Surinam. All instruments shaded against direct sunlight except one (after Schulz 1960).

logical (e.g. topsoil infiltration characteristics and erodibility) consequences have been discussed at length by Lal (1987).

However, an evaluation of any climatic changes brought about by forest removal requires a comparison of micro-climatic conditions over the newly cleared land with those prevailing at the top of the forest canopy rather than inside the forest (Thompson & Pinker 1975; Shuttleworth et al. 1985). As shown by Schulz (1960) and Thompson & Pinker (1975), temperatures and vapour pressure deficits experienced above a rain forest canopy are very similar to those recorded at screen height in large clearings (cf. Figure 13), although large differences may be found closer to the ground surface (Thompson & Pinker 1975; Pinker 1980).

There exists an important difference, however, in radiative characteristics exhibited by a rain forest canopy and various types of other surfaces. Tropical rain forests typically reflect about 12 per cent of the incoming short-wave radiation (Oguntoyinbo 1970; Pinker et al. 1980; Shuttleworth et al. 1984) whilst corresponding amounts for grassy clearings are close to 15 per cent (Pinker et al. 1980) or even 20 percent in the case of various agricultural crops (including well-watered grassland; Montény 1986). Therefore, a different partitioning of available energy between warming up of the boundary layer (i.e. sensible heat) and evaporation (i.e. latent heat) is to be expected upon conversion of tropical forest to grassland or agricultural crops, also because of differences in rooting depth and hence capacity to exploit soil moisture during dry spells (Shuttleworth 1988a). This in turn, if effected over a sufficiently large area, may affect local and regional circulation patterns of air and therefore rainfall (Salati & Vose 1984; Montény 1986; Dickinson & Henderson-Sellers 1988; Shukla et al. 1990; see below).

If, on the other hand, the forest is replaced by a vegetation type with rather similar radiative and evaporative characteristics, such as rubber or oil palm plantations, presumably the effect would be much less (Montény 1986). As such, it cannot be stressed enough that it is imperative to state the nature of "deforestation" when discussing these matters (Hamilton 1987).

The issue of the influence exerted by forest vegetation on the amount of rainfall it receives has already been touched upon in section 2.3.1. Two different approaches have been followed by investigators trying to resolve the question, which could be termed the "direct" and the "indirect" approach. The former analyses time series of rainfall and vegetation data whilst the latter uses computer simulation. Results obtained with both methods are reviewed below.

Circumstantial evidence for decreased rainfall at individual or a group of measuring stations in the tropics abounds in the literature and is often ascribed to concurrent "deforestation" (see numerous examples in Meher-Homji 1988). Unfortunately, many of these analyses cannot be considered very rigorous in that either the number of gauges or the period of observation taken into account were limited. In addition, synoptic considerations are rarely included in the analysis (cf. Dickinson 1980).

Mooley & Parthasarathy (1983) examined the occurrence of above- and below-average annual rainfalls between 1871 and 1980 for 306 stations all over the Indian subcontinent. They were unable to detect any trends or oscillations that were statistically significant and concluded that during the period under consideration annual rainfall totals over India were distributed randomly in time. Although it could be argued that most of the stations used in this analysis had lost their forest cover a long time ago (Meher-Homji 1988), the occurrence of a very wet or dry year seemed to be related to the degree to which depressions were able to penetrate the subcontinent in a westward direction. Wet years showed a distinctly higher proportion of depressions moving west of 80° E.L. (Mooley & Parthasarathy 1983).

In addition, the regional distribution of rainfall over India appeared to be strongly related to the location of the "monsoon trough", a zone of relatively low pressure which normally runs between South Bengal and Rajasthan (Ramaswamy 1962). The trough may shift towards the foothills of the Himalaya, producing a marked decrease in rainfall south of it (i.e. over the subcontinent) and a distinct increase in precipitation over the Himalaya (Dhar et al. 1982). These so-called "breaks" in the monsoon were shown to occur about three times more frequently during "drought" years than during "flood" years, whilst the average length of "break" was two to three times higher during dry years as well (Mooley & Parthasaraty 1983).

The above phenomenon could well explain a number of observations of locally decreased rainfall quoted by Meher-Homji (1988) but not such long-term persistent trends as that described for upland Sri Lanka by Madduma Bandara & Kuruppuarachchi (1988) (ca. 500 mm yr^{-1} between 1878 and 1970 in an area where a substantial portion of forest has been converted to tea plantations over the years). Incidently, Werner (1988), discussing the dieback of montane forest in the same area after a severe drought in 1976, reported that much higher rainfall totals have occurred again from 1984 onwards and that the forest was on the road to recovery. Since ET of mature tea plantations is not dramatically lower than that for montane forest (Blackie 1979b; Table 4), the above decrease in rainfall can hardly be attributed to the change in land use alone (Madduma Bandara & Kuruppuarachchi 1988). Rather, one must think of changes in the movements of the equatorial trough (cf. Arulanantham 1982).

Clearly, there is a need for rigorous statistical analysis of long-term rainfall records for carefully selected representative stations in relation to concurrent data on vegetation cover. Ideally such work should take into account synoptic situations as well. Arguably, data from areas with relatively high rates of sub-recent "deforestation" such as Ivory

Coast, Costa Rica, Thailand or Sumatra (Jackson 1983) could be used in such an analysis.

Fleming (1986) analysed time trends of annual rainfall totals for ten stations in Costa Rica with records ranging between 28 and 95 years. Lowland sites all exhibited a decrease in precipitation with time whereas virtually all stations in the hills showed an increase, although the statistical significance of the trends was weak. However, the slope of the regression line relating annual precipitation with time increased significantly with elevation. Fleming (1986) hypothesised that the large-scale conversion of semi-deciduous forest in the western lowlands of the country to pastures and dry-land cropping had brought about changes in air mass circulation that favoured the discharging of precipitation over the hills (cf. Meher-Homji 1980; Nooteboom 1987. It would be interesting to examine whether any long-term trends in spatial variations in sunshine duration exist in the area as well to see whether patterns of cloud cover (e.g. using satellite imagery) have changed accordingly.

Tangtham & Sutthipibul (1989) compared regionally averaged data on rainfall amounts and occurrence for thirty six stations in northeast Thailand with changes in forest cover over the period 1951-1984. On a year to year basis there was no correlation whatsoever between any rainfall parameter and the percentage of remaining forest cover, although annual rainfall totals generally exhibited a weak negative trend during the period under consideration. However, when annual rainfalls were expressed as 10-year moving averages there was a significant negative correlation with remaining forest area whilst a positive correlation was found between the latter and the number of rainy days. In other words, showers tended to become more frequent and smaller, although Tangtham & Sutthipibul (1988) were quick to point out that the "effect" of deforestation was still within one standard deviation of the means of the respective time series.

An opposite trend, i.e. less showers but of higher intensity, has been suggested by long-term observations of daily rainfall on private rubber estates in Peninsular Malaysia (UNESCO 1978). More work is needed.

The "indirect" approach (computer simulation of climatic effects of land-use changes) avoids the problem of high spatial and temporal variability in tropical rainfall to some extent but has its own share of problems.

Henderson-Sellers (1987) considered a realistic estimate of climatic effects of tropical deforestation "nearly impossible" for two main reasons: (1) the lack of reliable data on the nature and extent of "deforestation" and (2) limitations of the simulation models themselves and inadequacies in the statistical methodology for interpreting results obtained through them. She compared the results of four early simulations for the Amazon basin (Potter et al. 1975; Lettau et al. 1979; Henderson-Sellers & Gornitz 1984; Wilson (1984) in Henderson-Sellers 1987) and showed these to be rather contradictory.

For example, changes in temperature ranged from a decrease of about 0.5 °K through no change at all to an increase of about 0.5 °K, whereas changes in annual precipitation ranged from an increase of 75 mm (Lettau et al. 1979) to reductions of about 200 á 230 mm (Henderson-Sellers & Gornitz 1984; Potter et al. 1975) or 100 to 800 mm (depending on location; Wilson (1984) in Henderson-Sellers 1987). Although part of these discrepancies arose from differences in the nature of the conversion (the studies by Wilson and Henderson-Sellers & Gornitz related to a conversion to grassland), it was clear that at least the results obtained for changes in temperature did not match observed values (cf. Schulz 1960; Pinker 1980; Lawson et al. 1981; Luvall 1984). Henderson-Sellers concluded that both land surface parameterisations (LSP) and global circulation models (GCM) were in need of improvement.

Since Henderson-Sellers her essay was written (late 1984/early 1985) two sophisticated LSP's, namely BATS ("Biosphere-Atmosphere Transfer Scheme; Dickinson 1984; Henderson-Sellers et al. 1986; Wilson et al. 1987) and SiB ("Simple Biosphere" model; Sellers et al. 1986; Dorman & Sellers 1989) have become available and calibrated for tropical rain forest (Dickinson (1989) and Sellers et al. (1989) respectively). The calibrations were carried out using the extensive micro-meteorological and plant physiological data set available for the Ducke forest reserve in central Amazonia (Shuttleworth 1988a; Roberts et al. 1990) and showed the strong superiority of the improved LSP's over earlier over-simplified ones (Shuttleworth et al. 1990).

The on-site soil-vegetation-atmosphere models (i.e. the LSP's) may be appended to the base of a number of grid areas within a GCM and by changing the LSP for forest to, say, degraded grassland, the climatic effects of the conversion may be simulated. Two such simulations have been carried out recently for the Amazon basin (Dickinson & Henderson-Sellers 1988; Shukla et al. 1990).

The former study concentrated on the on-site effects of changes in albedo and/or surface roughness (forests create more turbulence than a smoother surface like grassland) on foliar and soil temperatures, latent and sensible heat fluxes, "runoff" and soil water over a range of monthly rainfalls. Air temperatures following the conversion typically rose by 1-3 °K and soil temperatures by 2-5 °K (Dickinson & Henderson-Sellers 1988) which must be considered as a major improvement compared to the results quoted earlier in this section, even though the inferred warming of the surface is model dependent.

Dickinson & Henderson-Sellers (1988) also found reduced evaporation (up to 50 per cent) and precipitation (ca. 20 per cent) as well as a lengthening of the dry season after conversion. However, since the GCM they employed showed weaknesses in the simulation of the extent and duration of convective cloud cover (Shuttleworth et al. 1990), these results should be viewed with caution for the time being.

The successful calibration of SiB for a tropical rain forest (Sellers et al. 1989) opened the possibility for a more realistic estimate of the climatic effect of a large-scale conversion of Amazonian forest to grassland. Using a GCM of relatively high resolution (Kinter et al. 1988) and synthesising an LSP for degraded grassland from the literature, Shukla et al. (1990) carried out a simulation over a 13-month period. The result of the exercise is summarised in Figure 14.

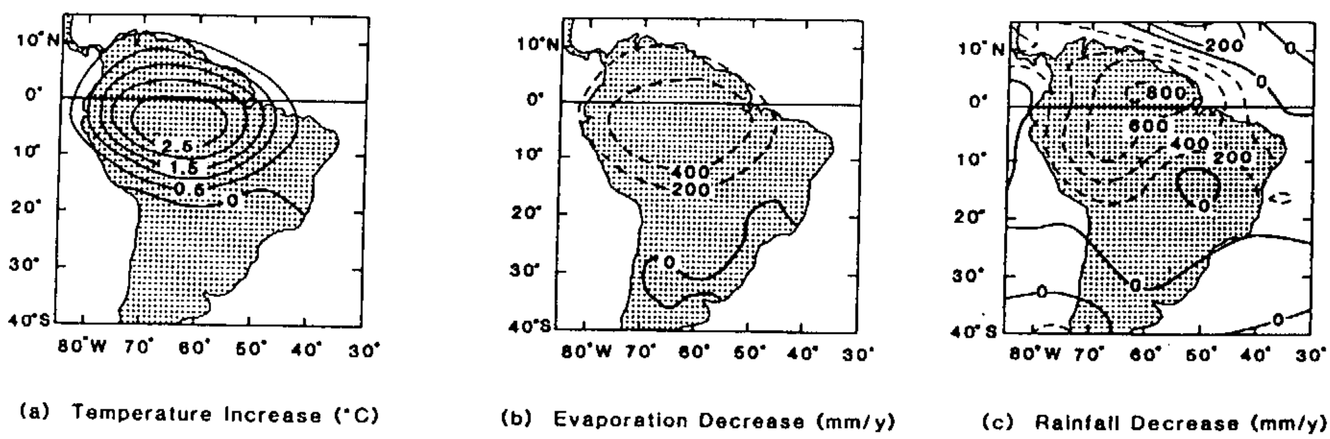

Figure 14. Predictions of the change in climate following a conversion of Amazonian rain forest to grassland (after Shukla et al. 1990).

Depending on location, predicted rises in air and soil temperatures were up to 2.5 and 3.5 °C respectively (as found by Dickinson & Henderson Sellers 1988), whereas basin wide rainfall and evapotranspiration were reduced by 26 and 30 per cent respectively (see Figure 14 for regional variations). The reduction in calculated annual precipitation by 640 mm and in ET by almost 500 mm suggests that changes in atmospheric circulation brought about by the conversion may act to further reduce the convergence of moisture flux in the Amazon region, a result that could not have been anticipated without the use of a dynamic model of the atmosphere (Shukla et al. 1990). For instance, Shuttleworth (1988a) predicted a drop in ET of about 20 per cent which was converted (on the basis of the recycling hypothesis; Salati et al. 1979) to a reduction in rainfall of about 10 per cent. Also, a reduction in ET could have been compensated for by an increase in moisture flux convergence.

The results of the experiment of Shukla et al. (1990) suggest that a compensation of this kind may not occur. Whether this is model-dependent will have to resolved by additional experiments and comparison with predictions by other models. If it is real, however, the ecological consequences will be enormous. As pointed out by Salati & Vose (1984) the presently experienced dry period in central Amazonia is the maximum the ecosystem can withstand. Any lengthening of the dry season will increase fire hazard (cf. Uhl & Buschbacher 1985) as well as influence a host of plant-animal interactions (Prance 1985, 1986b) which in turn could lead to irreversible changes in the vegetation.

Although the simulation by Shukla et al. (1990) is by far the most sophisticated of the attempts published to date, it should not be forgotten that the LSP for degraded grass- and scrubland employed in the study was based on data assembled from the literature rather than on actually determined values. Obviously, the use of other values for such important surface characteristics as reflection coefficient and soil water capacity might have produced different results (cf. Dickinson & Henderson-Sellers 1988). For example, in light of observations by Pinker et al. (1980) on the radiative properties of cleared areas occupied by grasses and scrubs in Thailand (13.4-14.3 per cent depending on season), the choice of a reflection coefficient of 21.6 per cent by Shukla et al. seems rather high. Similarly, Luvall (1984) reported net radiation totals over primary forest and a fresh clearing in Costa Rica to be very similar, although there was a profound change in the Bowen ratio (i.e. the partitioning of energy between sensible and latent heat). Therefore, the recently announced Anglo-Brazilian Amazonian Climate Observational Study (ABRACOS) which will study the water and energy dynamics of degraded Amazonian pastures (Shuttleworth et al. 1990) is a most welcome contribution in this respect and makes one look forward to its results. In addition, ABRACOS will address differences in micro-climate between large clearings and adjacent forest land in both the eastern (drier) and western (wetter) parts of Amazonia. In this way, it is hoped to improve the performance of GCM's in simulating extent and duration of convectional cloud cover as well as to improve the predictive ability of climate models in general (Shuttleworth et al. 1990; cf. Henderson-Sellers & Pitman 1990).

Summarising, great advances have been made in our understanding of forest-atmosphere interactions in the last few years. Although a number of questions remains to be answered (e.g. with respect to the maximum area that may be cleared without deleterious meso-scale climatic effects), it seems clear that the original contention of negligible influence put forward by Bernard (1945) and supported by Penman (1963) is no longer tenable in the light of more recent research results (Salati et al. 1979; Shukla et al. 1990).

4.3 Effect on water yield

Another common notion about the role of forests is that the complex of forest soils, roots and litter acts as a sponge soaking up water during rainy spells and releasing it evenly during dry periods. Although forest soils generally have higher infiltration and storage capacities than soils with less organic matter (Pritchett 1979), often much of this water is consumed again by the forest rather than used to sustain streamflow (cf. Table 1). Moreover, appreciable quantities of rainfall (up to 35 per cent; section 2.3.4) may be intercepted by the canopies of tropical forests and evaporated back into the atmosphere.

When dealing with the issue of effects of forest conversion on streamflow it is helpful to distinguish between effects on *water yield* (i.e. total streamflow) and on *flow regime* (the seasonal distribution of flow). The present section summarises what is known about the former whereas section 4.4 will discuss the latter aspect in more detail. Before examining the available information on effects of land use changes in the tropics on water yield a few methodological comments are necessary.

Simply comparing streamflow totals for catchment areas with contrasting land use types may lead to wrong conclusions because of the possibility of differences in catchment leakage (section 2.3.3). For example, Richardson (1982) found water yields for small catchments covered with montane rain forest and mature plantations of *Pinus caribaea* in Jamaica to differ by about 150 mm yr^{-1}. However, the corresponding values for ET were in excess of what has been reported for similar forests elsewhere by about 700 mm (Table 2), suggesting considerable catchment leakage. Similarly, flow from a 50-year-old plantation of *Eucalyptus robusta* in upland Madagascar was some 210 mm yr^{-1} below that for natural forest (Bailly et al. 1974; wet years only). Since the eucalypt covered catchment was much smaller than the rain forested one, and since small catchments in the area showed consistently lower streamflow totals than larger ones with the same vegetation (Bailly et al. 1974), it remains to be seen to what extent the quoted difference in flow reflects a real vegetation effect or rather a difference in catchment leakage.

Another complicating factor in the evaluation of hydrological effects of land cover transformations is the strong year to year variability of weather in the tropics (Qian 1983; Dyhr-Nielsen 1986). In addition, there is the large spatial variation in convective tropical rainfall which may render estimates of areal precipitation inputs for forested basins (which often have low rain gauge densities) relatively unreliable (Ribeny & Brown 1968; Aitken et al. 1972; De Bruin 1977).

An effective way to overcome some of these problems is the so-called "paired catchment method". Basically, the technique involves the hydrological comparison of two (or more) catchments of (preferably) similar size, geology, slopes, exposure and vegetation, situated close to one another: a "control" (to be left unchanged), and an "experimental" or "treatment" basin (Roche 1981; Hewlett & Fortson 1983).

The comparison is made during an initial calibration phase (which may

Plate 1. Scientific evidence is growing that evaporation from extensive tracts of rain forest produces part of its own rainfall.

Plate 2. "Cloud forest" at 2400 m on Mt. Kinabalu, Sabah, Malaysia.

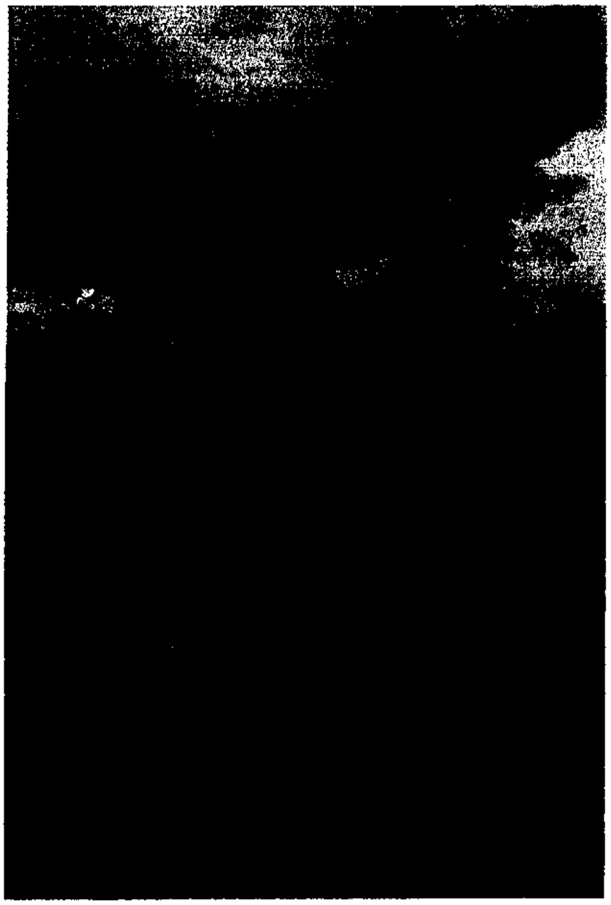

Plate 3.

Shifting cultivation in East Kalimantan, Indonesia (photo by K.F. Wiersum).

Plate 4. Rain forest that was selectively logged about fifteen years ago, Danum Valley, Sabah, Malaysia.

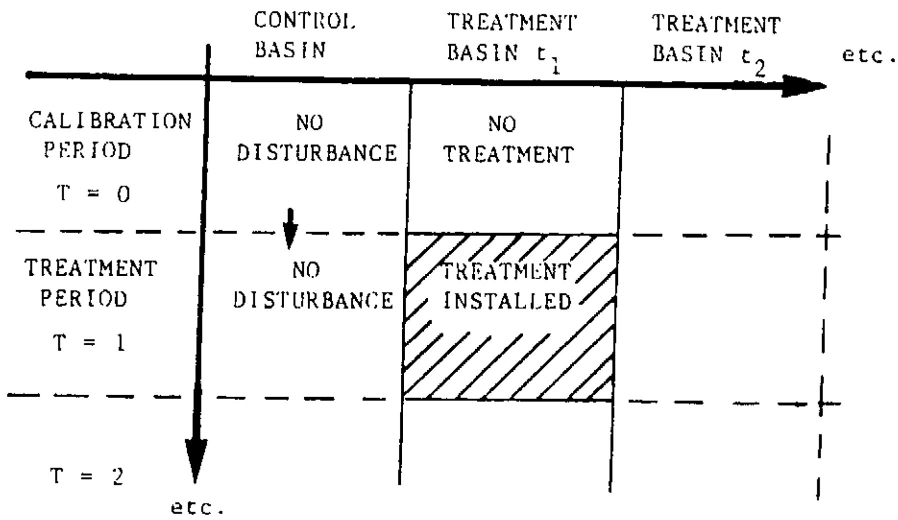

Figure 15. The paired catchment technique: general experimental design (after Hewlett & Fortson 1983).

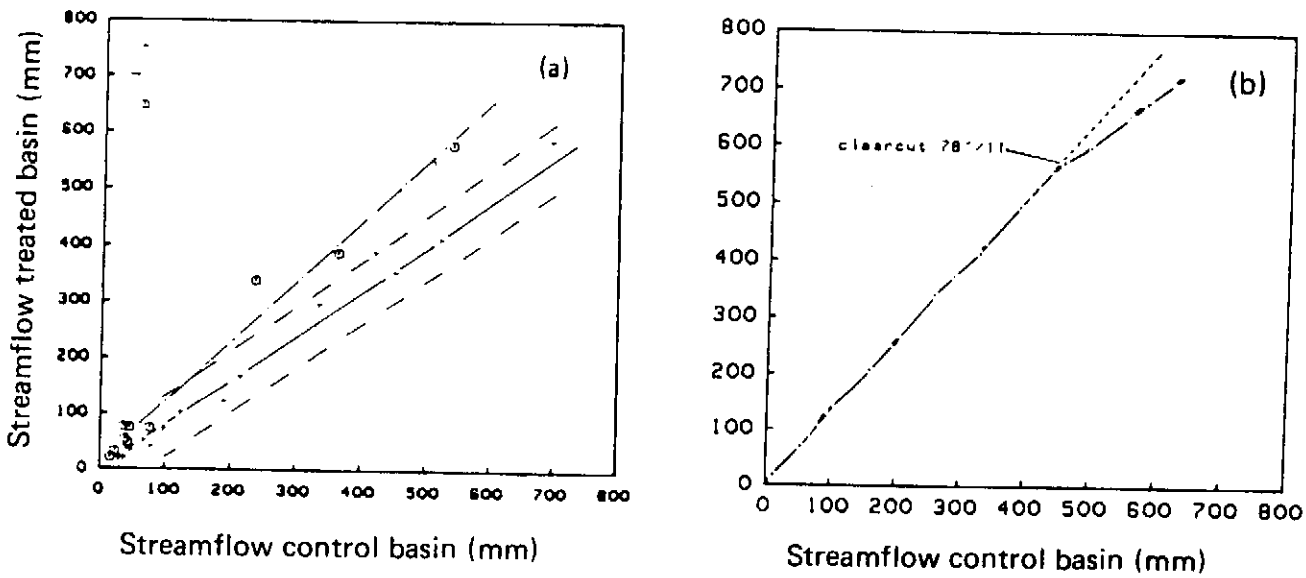

Figure 16. Evaluation of treatment effect by statistical analysis: (a) linear regression, (b) double mass curve (after Hsia & Koh 1983).

take several years, depending on rainfall variability (Kovner & Evans 1954), and during a subsequent treatment period in which the change in land use is effected (Figure 15). The degree to which linear regression equations (Figure 16a) or double mass curves (Figure 16b) linking the streamflows of the two catchments (as derived during the calibration period) change after the treatment is a measure of the effect of the latter.

The total duration of this type of experiment may easily span a decade in certain cases (calibration, clearing, site preparation, planting, maturation of the new vegetation; cf. footnotes Table 4). In addition, the results may be rather site specific due to an area's geological or pedological setting (Fritsch et al. 1987; Dano 1990). Therefore, in recent years there has been an increasing trend to predict hydrological changes brought about by land cover transformations in the tropics by robust models employing data obtained during relatively short (up to two years) but intensive measuring periods (Vugts & Bruijnzeel 1988; Shuttleworth 1988a; Shuttleworth et al. 1990; Institute of Hydrology 1990).

Bosch & Hewlett (1982) reviewed the results of almost hundred paired basin experiments throughout the world including a few from the tropics to determine the effect of vegetation removal or modification on basin water yield. They concluded that "no experiments in deliberately reducing vegetation cover caused reductions in water yield, nor have any deliberate increases in cover caused increases in yield".

In other words, *removal of forest cover leads to higher streamflow totals and reforestation of open lands generally leads to a decline in overall streamflow.*

Table 4 summarises the presently published data with respect to the effect of land cover transformation on water yield in the humid tropics, including a few examples from somewhat more seasonal subtropical sites. Most of the data are based on paired catchment experiments (see footnotes for details).

Results have been listed according to site elevation, starting with low intensity disturbances in lowland forests (no's 1-3), followed by various types of conversion in lowland areas (no's 4-7) and in more elevated terrain (no's 8-15). The last five studies (no's 16 - 20) deal with effects of reforestation, burning and coppicing.

The following conclusions may be drawn from Table 4:

(1) Carefully executed light selective harvesting will have little (if any) effect on streamflow (no's 2 and 9), whilst the effect increases with the amount of timber removed (case study no. 1);

(2) The data set for the humid tropics supports the general finding of Bosch & Hewlett (1982) that removal of the natural forest cover may result in a considerable initial increase in water yield (up to 800 mm yr^{-1}; possibly more in high rainfall regions: no. 4), depending mainly on the amount of rain received after the treatment (no's 1, 2, 5, 8, 12);

(3) Depending on rainfall patterns, there is a rather irregular decline in streamflow gain with time associated with the establishment of the new cover (no's 2, 5, 8); no data have been published regarding the number of years needed for a return to pre-cut streamflow totals in the case of natural regrowth; accepting the results of no. 11 at face value, this may take more than eight years; more work is needed;

(4) Water yield after maturation of the new vegetation may remain above original streamflow totals in the case of conversion to annual cropping (no's 7, 14), grassland (no's 4, 6, 12, 15 with larger increases under wetter conditions) or tea plantations (no. 13), return to original levels (pine plantation after full canopy closure: no. 14) or remain below previous values (reforestation of grassland with pines or eucalypts: no's 18-20); coppicing of eucalypts after ten years caused even stronger reductions for two years (no. 18);

(5) Burning of grassland may increase flow (no. 19, *Loudetia, Aristida*) or reduce it (no. 15, *Imperata*); in the former case this was mainly due to an increase in stormflow (burning every other year), in the latter it may be related to enhanced water uptake during and after renewal of above-ground biomass (annual burning); further investigation is desirable.

TABLE 4. Changes in annual water yield associated with changes in land cover in the (sub)tropics

Location	Type of transformation	Method	Change in water yield (mm yr^{-1}) 1st year	2nd year	3rd year	4th year	nth yr	
(1) Bukit Berembun, Malaysia[1-3]	Commercial selective logging (catchment C1, 40 % removal)	PCT	165^{w} (70%)	140 (53%)	175^{w} (72%)			
	Supervised selective logging (catchment C3, 33 % removal		85^{w} (37%)	70 (28%)	105^{w} (44%)			
(2) Babinda, Queensland[4]	Selective logging (N Creek)	PCT	effect not statistically significant					
	Clearing 67 % of basin and subsequent regeneration		265 (7%)	325^{w} (13.4%)				
(3) La Selva, Costa Rica[5]	Manual clearing (no burn) and subsequent regrowth	P-M	375* (26%)					
(4) St. Elie, Fr. Guyana	Rain forest to eucalypt plantation[6-8]	PCT	410/825*	330/680*	40/185*			
	Idem to <u>Pinus caribaea</u>[7,8]		495/925*	380/725*	210/435*			
	Idem to <u>Digitaria</u> grass on poorly drained soil[9]		average over four years: 235/270*					
	Idem to <u>Brachiaria</u> grass on freely drained soil[9]		average over four years: 230/325*					
(5) Sungei Tekam, Malaya[10-15]	Rain forest to cocoa	PCT	110^{d} (117%)	706^{w} (157%)	353^{w} (94%)	263 (158%)		
	Rain forest to oil palm (60 %; remaining 40 % cleared in 3rd/4th yr		145 (85%)	155^{d} (142%)	137^{d} (97%)	822^{w} (470%)	793^{w} (270%)	476 (314%)

(6)	Yangambi, Congo[16]	Secondary forest versus	SWB				
		- Paspalum grassland					60* (14%)
		- Bare soil					285* (66%)
(7)	IITA, Nigeria[17,18]	Clearing secondary forest for traditional cropping	CWB	305*w	170*	140*	
(8)	Lien-Hua-Chi, Taiwan[19,20]	Clearing of mixed evergreen hill forest, regeneration cut in 2nd year	PCT	450 (58%)	205d (51%)		
(9)	Rajpur, India[21]	20 % thinning of Shorea forest	PCT	no detectable change in first two years			
(10)	Blue Mnts., Jamaica[22]	LMRF vs. mature Pinus caribaea plantation	CWB				150*
(11)	Périnet, Madagascar[23]	Montane forest vs. sec. bush	CWB				200*
		Idem vs. mature Eucalyptus					-210*
(12)	Luano, Zambia[24]	Miombo woodland to cropping (ca. 10%) and grazing (2/3)	PCT	average increase over five years: 195 - 230 (56 - 74 %)			
(13)	Kericho, Kenya[25]	Montane forest to tea (54%)	PCT				
		I clearing/planting		three-year average: 220			
		II establishment		four-year average : -100d			
		III maturation		six-year average : 150			
(14)	Kimakia, Kenya[26]	Montane forest (bamboo) to plantation of Pinus patula	CWB				
		I intercropping phase		three-year average:			125
		II establishment		six-year average :			75
		III maturation (closed canopy)		seven-year average:			5

1-25, *see footnotes; PCT = paired catchment technique; P-M = Penman-Monteith model; SWB, CWB = site, catchment water balance; w,d = wet, dry year or period

TABLE 4 continued

Location	Type of transformation	Method	Change in water yield ($mm\ yr^{-1}$) 1st year	2nd year	3rd year	4th year	nth yr
(15) Mbeya, Tanzania[27]	Montane forest (67%), scrub (33%) vs. traditional agriculture (50% crop, 50% grass)	CWB	ten-year average :				410*
(16) Angat, Philippines[28,29]	*Imperata* grassland to fire protected grassland	PCT	four-year average :				120 (9.5 %)
	Idem to *Gmelina arborea*		six-year average :				80 (9.4 %)
(17) Pidekso, Indonesia[30]	Degraded cropland vs. mixed plantation forest (11 yr)	CWB					-580
(18) Dehra Dun, India[31-34]	Scrubland to *Eucalyptus*	PCT	five-year average:				-15 (28 %)
	Coppicing of eucalypts after ten years		-160^{d} (68%)	-285 (47%)	+10 (2%)		
(19) Manankazo, Madagascar[35]	Natural grassland to burned grassland	CWB/ SWB	11/6-yr-averages: 100 / 125				
	Idem to agriculture		11-yr average : - 75				
	Idem to *Pinus patula*		11-yr average : - 80 / 160 (6-yr-old trees)				
(20) Ootacamund, India[36]	Natural grassland to *Eucalyptus globulus* (59%)	PCT	average for first three years :				-10 (2 %)
			average between 4th and 10th yr:				-120 (21 %)

[26-36] see footnotes; SWB, CWB = site, catchment water balance; PCT = paired catchment technique; d = dry year

Table 4 continued (footnotes)

[1]Abdul Rahim & Baharuddin (1986); [2]Abdul Rahim (1989); [3]Abdul Rahim (1990); Experiment initiated in 1979; 3 adjacent rain forested basins at 170-300 m a.s.l.; deep Ultisols; granite; mean P ca. 1900 mm without real dry season; after 3 yr calibration against control basin (4.6 ha) commercial logging (40% stocking removed; 1.86 km of road/track - 0.14 km ha^{-1};) of basin C1 (13.3 ha) and supervised logging (30% removed; 2x20 m riparian buffer zone; 2.2 km road/track - 0.10 km ha^{-1}) of basin C3 (30.8 ha); crawler tractors and winch lorries.

[4]Gilmour (1977b);
Experiment initiated in 1969; mesophyll vine forest at 10-200 m; deep Ultisols with permeability rapidly decreasing with depth; metamorphic rock; wet with no mo with P <100 mm; after 18 mo of calibration against 25.7 ha control North creek basin (18.3 ha) selectively logged in 1971, 67% cleared and ploughed in 1973, then regeneration; area cleared again in December 1987 (M. Bonell, personal communication).

[5]Luvall (1984);
One-yr experiment; rain forest at 150-180 m; deep volcanic soils; P ca. 4000 mm without real dry season; ET via Penman-Monteith formula for primary forest and manually cleared 50x50 m block during 161 days; no removal or burning of slash; climatic data above-canopy except for wind; aerodynamic resistance computed from 10 day measurements of wind speeds at two levels above low vegetation; quoted value for 1st yr regrowth based on actual values for 1st half and taken as 80% of forest ET during 2nd half; since forest ET overestimated, results normalised to mean ET of Table 1.

[6]Fritsch (1983); [7]Fritsch & Sarrailh (1986); [8]J.M. Fritsch, personal communication; [9]Fritsch (1987); Large experiment initiated in 1977 lowland rain forest; soils with highly variable drainage; schists; climate wet and slightly seasonal; ten catchments (1-1.5 ha) calibrated for 17 (conversion to grassland) to 30 (conversion to plantations) mo before logging with heavy machinery; 1st values quoted refer to quickflow (Qq) and are given directly in [6-9], 2nd ones pertain to total runoff (Qt) computed from scattered data on Qq and Qt in [6-9] and are approximate; eucalypts (species not mentioned) exhibited very poor growth even after 2nd planting[8].

[10-11]DID (1975,1982,1986,1989); [14-15] Abdul Rahim (1987,1988); Experiment initiated in 1973, data available since 1977; lowland rain forest at 70 m; P ca. 1880 mm without real dry season; deep Oxisols on lateritic shales (control basin C, 56.2 ha) or on andesitic rocks and tuffs (experimental basins A and B); A (37.7 ha) headwater area of sub-B (59.2 ha), together making up B (96.9 ha); after 3 yr calibration sub-B (60% of B) logged and burned 2nd half '80; planting of cover crop spring '81 (= 1st yr); road construction and stream realignment in 2nd yr; planting of oil palm beginning of 3rd yr (late '82);

basin A (40% of B) logged and cleared between late '82 and mid '83 after 5 yr calibration; road construction and cover crop planting in 2nd half of '83 (= 2nd yr); cocoa planted late '83 and early '84, i.e. ca. 1 1/2 yr after planting of oil palm in sub-B.

[6]Focan & Fripiat (1953);
Secondary evergreen seasonal forest, Paspalum grassland and bare soil at 470 m; sandy clay soils; P ca. 1860 mm; seasonal climate but no details given; site water balance (cf. footnote 10 to Table 1 for details); values appear realistic.

[7]Lawson et al. (1981); [18]Lal (1983);
Experiment initiated in 1974; immature (?) secondary seasonal forest; Alfisol; P ca. 1450 mm with two dry seasons; CWB for single basin of 44 ha before and after clearing; no calibration; quoted values read from graph in [18] and 30 mm subtracted for flow in forested state[17]; values approximate.

[9]Hsia & Koh (1983); [20]Hsia (1987);
Study initiated in 1975; evergreen mixed broadleaf and coniferous forest at 725-785 m; fine silt loam soil; sandstone and shale; P 2100 mm, 80% between May and September; after 7 yr calibration against 5.9 ha control area, experimental basin (8.4 ha) cleared by skyline logging and uphill yarding; road around basin perimeter; minimal disturbance of soil; regrowth in 2nd yr cut for planting of China fir.

[1]Subba Rao et al. (1985);
Study initiated in 1964; dense Shorea forest at 895 m; deep sandy soils; P 2950 mm, mostly between June and October; flow not perennial; after 10 yr calibration against 6.5 ha control, experimental basin (5.2 ha) subjected to 20% thinning; measurements continued for 5 yr but only 1st 2 yr discussed.

[2]Richardson (1982);
One-year comparison of CWB for 38.5 ha basin with LMRF and 8.8 ha basin with 19-yr-old pines; no calibration; P 3750 mm with no real dry season; gravelly sandy loam; porphyrites and conglomerates; very high ET probably due to catchment leakage; quoted difference not reliable.

[3]Bailly et al. (1974);
Experiment initiated in 1962; long-term comparison of CWB for 38 ha basin with evergreen montane forest (D3) with that for 31.5 ha basin with secondary scrub (D7) and for 13.3 ha basin (D5) with 50-yr-old eucalypts; forest selectively logged 50 yr ago, plantation thinned just before start of experiment; vigorous undergrowth in D5; 930-1095 m; Ultisols; gneiss; P variable (1600-2100 mm) with 7 mo P<100 mm (2 <60 mm); all basins had excellent instrumentation; P at D5 some 500 mm below that for D7 and D3; quoted difference in Q based on average runoff coeffients and P of 2000 mm; difference between forest and scrub seems high but feasible, that between forest and eucalypts may be affected by leakage from D5.

[24] Mumeka (1986);
Experiment initiated in 1964; mixed Miombo (semi-open) woodland on freely drained soil surrounding seasonally inundated grasslan (dambo) on poorly drained soil; 1300 m; P 1400 mm; strongly seasonal (no details given); three excellently instrumented basins (95-143 ha) intercalibrated for 8 yr; 2 basins (A and G) cleared except for 60 m wide buffer strip around dambo; 10 ha used for improved agriculture, rest used as pasture.

[25] Blackie (1979a);
Experiment initiated in 1958; montane rain forest at 2200 m; dee volcanic soils; phonolitic lava; P 2130 mm with two months P<100 mm; after 18 mo calibration against 544 ha control basin (Lagan) 54 % of Sambret basin (702 ha) was cleared to establish tea plan ation; see footnotes Table 2 for instrumental details.

[26] Blackie (1979b);
Experiment initiated in 1958; montane rain forest at 2440 m; dee volcanic soils; pyroclasts over basalts weathered into dense cla P 2300 mm; CWB for forested basin C (64.9 ha) and for 36.4 ha ba A planted with pines just before observations; no calibration; basins believed watertight; see footnotes Table 2 for instrument details.

[27] Edwards (1979);
Experiment initiated in 1958; montane evergreen forest at 2500 m; one metre of volcanic soil over deeply weathered gneiss; P 1925 r with 6 mo with P<60 mm (4 completely dry); CWB for forested catch ment (C) of 16.3 ha and agricultural basin (A) of 20.2 ha; no cal ibration and some leakage (cf. high ET forest in Table 2) cannot excluded; no soil conservation measures on cropland in A; see footnote Table 2 for instrumental details.

[28] Dano (1990); [29] A.M. Dano, personal communication;
Experiment initiated in 1973; Imperata grassland at 225 m; clay loam soils; andesitic rock?; P 3170 mm with 3 mo P <60 mm; after 4 yr calibration against 1.6 ha control, Left Fork basin (0.95 ha protected against annual burning for 4 yr (1977-1980), then plant with Gmelina (poor growth, data until 1986[29]); burning continued control; only small part of Qt recorded which may have affected quoted values; highest increases during dry years and vice versa.

[30] Hardjono (1980);
One-yr comparison of CWB for 354 ha basin with mixed 11-yr-old pine, teak and mahogany plantations and 207 ha basin with dry-land agriculture; seriously degraded volcanic terrain; 375 m; thin soils; P 2600 mm with 4 mo of P <60 mm; no calibration; quoted values most probably affected by differences in basin leakage (cf. section 4.4.2).

[31]Mathur et al. (1976); [32]Mathur & Sajwan (1978); [33,34]Vishwanatham et al. (1980, 1982);
Experiment initiated in 1961; secondary scrub (5 yr old at start of observations) at ca. 520 m; stabilised gullies in undulating terrain; silty (clay) loam soils; P ca. 1430 mm, mostly between June and October; flow not perennial; quoted data refer to Qq; after 8 yr calibration against 0.87 ha control, basin 2 (1.45 ha) planted with _Eucalyptus grandis_ and _E. camaldulensis_ (2x2 m); results given as means for 5 yr after treatment; post-treatment included 2 very dry years.

[35]Bailly et al. (1974);
Experiment initiated in 1962; long-term comparison of CWB for 3.2 ha basin with natural grassland with that for 4.8 ha basin with grassland burned every other year, for 3.2 ha basin with improved dry-land agriculture and for 3.9 ha basin with newly planted pine; 1550 m; Ultisols; gneiss; P 1715 mm with 7 mo of P <60 mm; values refer to Qq.

[36]Samraj et al. (1988);
Experiment initiated in 1968; natural grassland at 2035 m with scattered stunted evergreen trees and swampy valley bottoms; deep permeable soils; P 1535 mm, concentrated between June and September; after 4 yr calibration against 33.2 ha control basin, 59% of catchment B (31.9 ha) planted with bluegum (2x2 m); trees grown in 10-yr rotation.

The almost complete lack of rigorous (i.e. involving basin calibration) studies dealing with the impacts of shifting cultivation or conversion of natural forest to annual cropping on streamflow is somewhat surprising since these two activities account for a major portion of tropical forest destruction (Myers 1980; Seiler & Crutzen 1980). Virtually all material deals with effects on infiltration capacity, surface runoff and on-site erosion (see section 4.5).

Bailly et al. (1974) compared amounts of flow emanating from small (1.4-1.8 ha) basins in upland Madagascar under old secondary vegetation, dry land agriculture with soil conservation measures and under temporary slash and burn agriculture (two years followed by regeneration), respectively. Since the small basins were leaking severely, results pertained to stormflow only. Runoff was consistently highest from the third basin and lowest from the first one. However, it is difficult to envisage how runoff from six-year-old regrowth may be more than twice as high as for old scrubland as reported by Bailly et al. (1974). Again, it cannot be excluded that differences in catchment characteristics have influenced the results (see also section 4.4.1).

Changes in flow from a catchment in Nigeria (no. 7 in Table 4) whose secondary forest cover was converted to dry land agriculture as reported by Lal (1983) were a little contradictory. On the one hand, surface runoff gradually increased with time after clearing, supposedly as a result of deteriorating infiltration opportunities. The associated decrease in soil water storage would have led to reduced baseflows but instead these were reported to increase over the three year period (Lal 1983). Since the experiment did not include a calibration of the catchment it is likely that the reported trends were influenced by variations in rainfall.

Edwards (1979) presented a long-term comparison of streamflow from two small and presumably watertight basins in upland Tanzania, one with evergreen forest and one converted to (traditional) agriculture just before the start of the observations (no. 15 in Table 4). A consistent difference in water yield of about 400 mm yr^{-1} was observed during the ten year period of measurement. Incidently, throughout the entire duration of the experiment no environmental deterioration occurred in the cleared catchment, despite the fact that no soil conservation measures other than the bunding of harvest residues were applied, much to the surprise of the investigator himself (Edwards 1979). This rather unexpected result could be explained by a combination of relatively stable volcanic soils capable of maintaining high infiltration capacities (cf. Lungren 1980) and low intensity rainfalls.

Although circumstances at Mbeya are atypical for the humid tropics and these results must therefore be considered exceptional, they nevertheless illustrate the potential for watershed management in the tropics through modification of the surface cover *in conjunction with soil conservation measures* (cf. Bailly et al. 1974). Much is expected from agroforestry in this respect (Nair 1984; Ewel 1986; Young 1986; Vandermeer 1989; Bonell 1989) although little information is available as yet about the water dynamics of the various agroforestry systems.

Imbach et al. (1989) computed water balances for two such systems in Costa Rica, viz. cocoa with *Erythrina poeppigiana* and *Cordia alliodora* as the respective shade trees, using a somewhat insensitive method for the evaluation of transpiration and hence percolation. Taking their estimates of ET (800 and 1025 mm yr^{-1} respectively) at face value, permanent gains in water yield after conversion of upland forest to these types of agroforestry of about 200 and 400 mm could be expected (cf. Table 2).

Large tracts of lowland forest have been (and are still being) converted to extractive tree plantations such as oil palm and rubber in such rapidly developing countries as Nigeria, Ivory Coast or Malaysia (Bertrand 1983; Abdul Rahim 1985; Lal 1987) before the long-term hydrological consequences were known very well. As indicated before, paired catchment experiments studying the effects of conversion of tropical forest to tall vegetation take a long time to complete. Therefore, most of the studies quoted in Table 4 have covered only the early phase of such a conversion.

To answer the question whether water consumption of fast growing and regularly fertilised extractive tree crops in their mature phase will exceed that of the natural forest they have replaced, requires additional information.

As for oil palm, lysimetric studies on a young tree by Ling (1979) in Malaysia confirmed the initial observations of the paired catchment study at Sungei Tekam (no. 5 in Table 4), i.e. a reduction in water use compared to rain forest of 200-250 mm yr^{-1}. Later work by Foong et al. (1983) using the same large lysimeter showed an increase in annual ET to about

1800 mm for a regularly irrigated four- to six-year-old palm tree. Using crop factors derived by Wormer & Ochs (1959) and assuming a (low) soil water capacity value of 100 mm for the 140 cm soil column of the lysimeter, Foong et al. (1983) estimated an ET of about 1450 mm yr^{-1} for a mature non-irrigated oil palm plantation. As we have seen (Table 1), the latter value is very similar to that for natural lowland forest. To the extent that the above estimate is realistic, streamflow levels in terrain with mature non-irrigated oil palm plantations may be expected to be similar to those experienced before the conversion.

As for rubber, Haridas (1985) reported the very high value of 1700 mm yr^{-1} for the water use of a young (up to six years old) stand in Malaysia. The streamflow component of his water balance was based on instantaneous readings of water level taken twice a day rather than on continuous recording. In addition, his estimates of changes in annual soil water storage (no details on methodology) were unrealistically high (620-1605 mm) under the prevailing climatic conditions. As such, the estimate presented by Haridas (1985) must be considered suspect. The average annual deficit (i.e. rainfall minus streamflow) amounted to 490 mm.

Montény et al. (1985) reported values of 920 to 990 mm yr^{-1} for a mature (19 years old) stand of rubber under more seasonal conditions in the Ivory Coast. Their estimate was based on an excellent set of above-canopy climatic data. Adopting the higher value of the two as representative for mature rubber plantations, then a conversion from natural forest to rubber would indeed imply a permanent increase in water yield of 100-300 mm yr^{-1} (Montény et al. 1985; Montény 1986; cf. Table 1).

In view of the large extent of degraded grassland in the tropics (Thijsse 1977; Dano 1990) their forestation with fast growing tree species such as pines and eucalypts is likely to become increasingly important (Hamilton & Pearce 1987; Postel & Heise 1988). The results presented in Table 4 as well as several studies in the sub-humid part of Fiji (Kammer & Raj 1979) and in South Africa (Bosch 1979; Van Lill et al. 1980) suggest a decline in water yield following forestation of degraded grass- or cropland with pines or eucalypts.

These reductions in flow are caused by the increased evaporative loss from tall vegetation during rainy spells (interception; section 2.3.4) and by the higher transpiration from trees during dry periods, reflecting the generally more extended root network of trees as compared to short vegetation (Eeles 1979; Calder 1982).

At first sight the results obtained by Dano for forestation of annually burned *Imperata* grassland with *Gmelina arborea* in the Philippines (no. 16 in Table 4), viz. an increase in flow over the first six years of 80 mm yr^{-1}, seem to contradict the above statements of decreased water yield following forestation of grassland. However, it should be remembered that the control basin was burned every year whilst protecting the grass from fire during an earlier phase of the experiment had produced the same relative increase in flow as forestation (Table 4). As such, it is difficult to distinguish between the effects of the two factors. Also, the plantation suffered from browsing deer and growth was rather poor (A.M. Dano, personal communication).

Whether replacement of tall natural forest by eucalypts will also produce a decline in streamflow remains to be seen. The reduced water yield recorded for a small upland basin covered with mature *Eucalyptus robusta* in Madagascar as compared to that for nearby catchments with natural forest (Bailly et al. 1974; site no. 11 in Table 4) has already been commented on in terms of possible differences in catchment leakage.

Calder (1986) reviewed the literature on water use by eucalypts. His conclusions that (1) interception losses from eucalypts were generally less than those from other tree species of similar height and planting density, and (2) transpiration rates from eucalypts were likely to be similar to other tree species except in situations with a shallow groundwater table, were recently substantiated by the results of an elaborate measuring programme in subhumid South India (Institute of Hydrology 1990).

Therefore, the widespread fear that eucalypts are "voracious consumers" of water (Vandana Shiva et al. 1982; Vandana Shiva & Bandhyopadhyay 1983) is not supported by recent research (Institute of Hydrology 1990). Nevertheless, additional work in perhumid climates would be welcome.

The results presented thus far all pertain to relatively small catchment areas involving a unidirectional change in cover. Although these provide a clear and consistent picture of increased water yield following a replacement of tall vegetation by a shorter one and vice versa, effects of conversion may be more difficult to discern in the case of larger basins having a variety of land use types and vegetation in various stages of regeneration. In addition, there is the strong spatial and temporal variation in tropical rainfall referred to already which tends to "drown" any effects associated with modifications of the surface.

For example, Qian (1983) was unable to detect systematic changes in streamflow patterns of basins ranging in size from 7 to 727 km^2 on the island of Hainan, South China, during the 1960's and 1970's despite extensive "deforestation" (ca. 30 per cent loss of tall forest between 1950 and 1980). Dyhr-Nielsen (1986) arrived at the same conclusion after studying rainfall and runoff data over the period 1955-1980 for the 14500 km^2 Pasak river basin in northern Thailand which lost up to 50 per cent of its tall forest cover during this period.

Richey et al. (1989) reconstructed an 83-year record of the discharge of the Amazon river at Manaus and showed that there had been no statistically significant change in discharge over the period of record (1903-1985) which predates significant human influences in the river basin. Interestingly, cross-spectrum analyses of Amazon flow anomalies with indicators of the El Nino southern oscillation phenomenon (notably atmospheric pressure anomalies at Darwin, northern Australia, and monthly mean sea surface temperature in the eastern and central equatorial Pacific) suggested that the oscillations in the hydrograph were coupled to the tropical Pacific climate cycle on a two- to three-year time scale. Richey et al. (1989) refuted the assertion of Gentry & Lopez-Parodi (1980) that the increases in flow recorded between 1962 and 1978 were caused by deforestation in the Andean uplands and stressed that long-term records should be examined before ascribing short-term "anomalies" to anthropogenic influences.

However, detectable changes in streamflow may be expected in such cases where large tracts of forest vegetation have been replaced by annual cropping rather than left to regenerate. Odemerho (1984a,b) observed significant changes in stream channel geometry (i.e. channel widening) in headwater basins in southwestern Nigeria following a shift in land use from slash and burn agriculture with long fallows through one with short fallows to permanent cultivation.

Similarly, Madduma Bandara & Kuruppuarachchi (1988) reported an increase in mean annual flow for the 1108 km^2 Mahaweli basin in Sri Lanka despite a weak negative trend in basin rainfall over the same period

(1944-1981). Although both trends were not statistically significant at the 95 per cent level, the increase in annual runoff ratio's was highly significant (Figure 17). Since the upper half of the basin in which relatively little change in land use had taken place exhibited a negative trend in annual discharge reflecting that in rainfall, the increased hydrological response of the lower half of the basin must have been caused by the widespread conversion of tea plantations to annual cropping and home gardens (Madduma Bandara & Kuruppuarachchi 1988).

In many parts of the tropics, especially in areas experiencing a dry season, the seasonal distribution of streamflow assumes greater importance than total annual water yield. The next two sections examine the evidence available to date with respect to the influence of cover transformations on peak flows ("floods") and dry-season flow respectively.

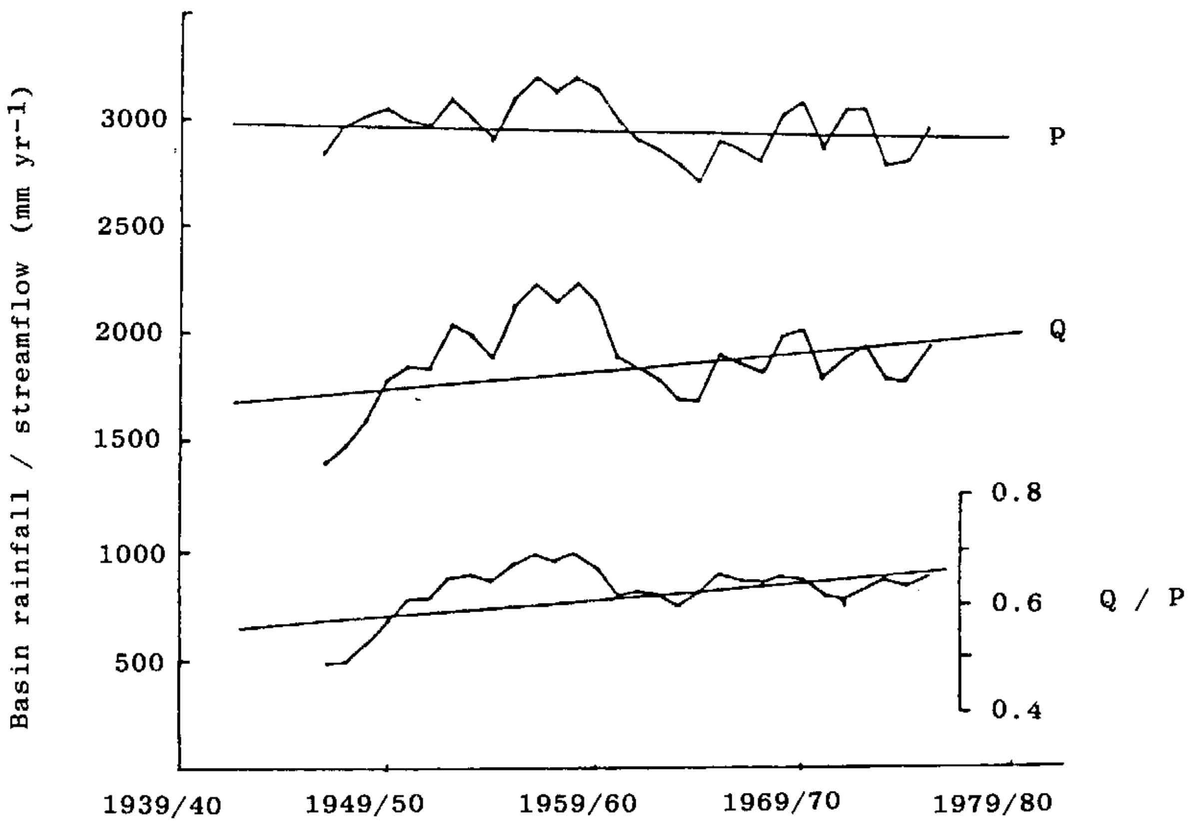

Figure 17. Five-year moving averages of rainfall, streamflow and runoff ratio's in the upper Mahaweli basin above Peradeniya, Sri Lanka (after Madduma Bandara & Kuruppuarachchi 1988).

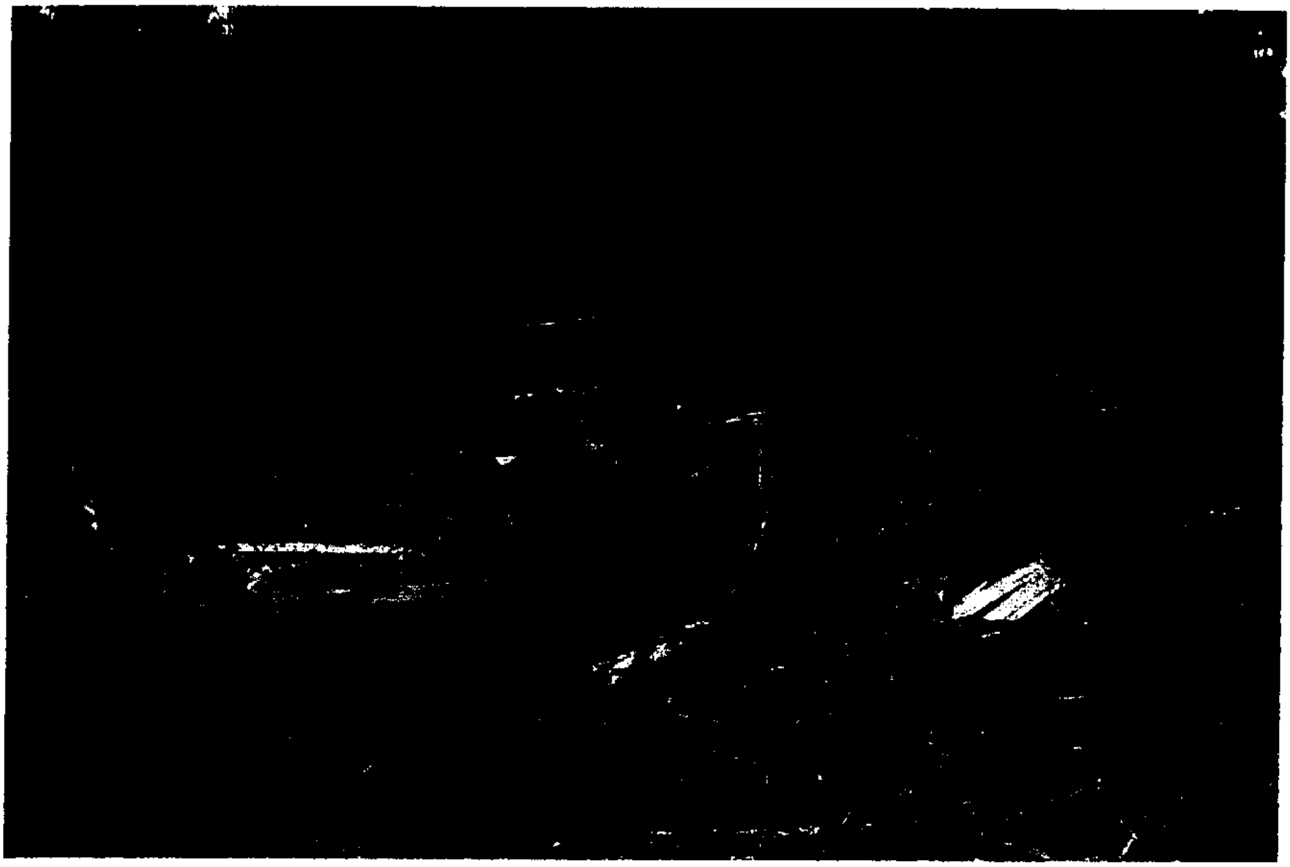

Plate 5. The use of heavy equipment to extract timber involves considerable local disturbance of the soil surface.

Plate 6.

Logging roads and tractor tracks may become important sources of runoff and sediment (photo by Anders Malmer).

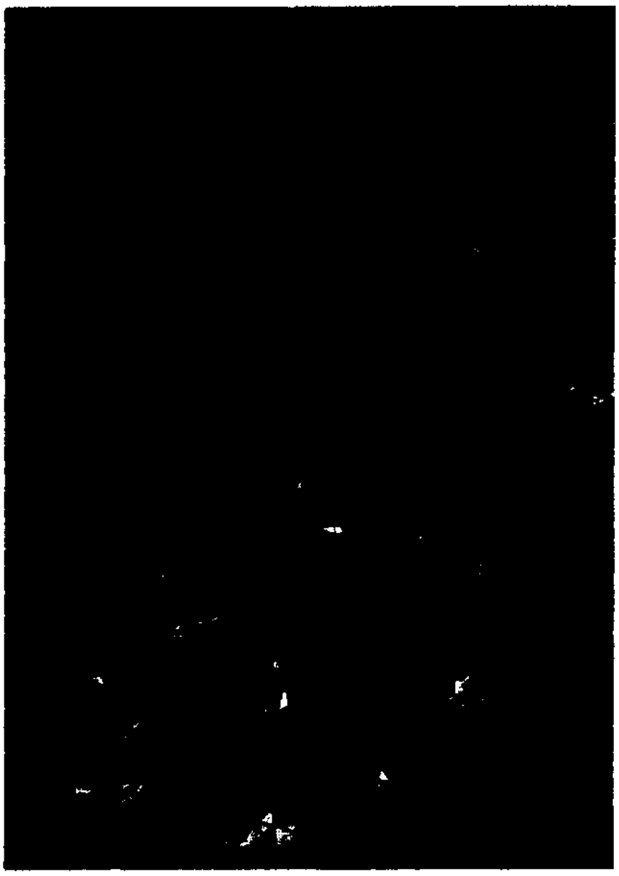

Plate 7.

Surfaces that became heavily compacted during logging often exhibit poor regeneration. Photograph taken 15 years after logging, Segama area, Sabah.

Plate 8. Skyline logging may provide an alternative in steep terrain where surface disturbance is to be kept to a minimum, Segama area, Sabah.

4.4 Effects on streamflow regime

4.4.1 Floods

As shown in section 2.1, the hydrological response to rainfall of small basins depends on the interplay between climatic, geological and land use variables (Figure 5). Peaks pronounced by some form of overland flow, be it the infiltration-excess / Hortonian type (HOF) or saturation overland flow (SOF), are generally much more pronounced than those generated by subsurface types of flow (Figure 4). It is especially the shift from subsurface flow- to HOF-dominated stormflows which often accompanies certain changes in land use, that may produce rather dramatic changes in basin hydrological response (Table 5).

However, caution must be applied when simply comparing results for different catchments with contrasting land uses since geological and topographical (e.g. basin shape) factors may override the vegetation effect (Roessel 1950). For example, storm runoff from a 70-ha forested basin in the Lesser Himalaya was consistently higher than for a nearby agricultural basin of 55 ha (Sastry et al. 1983; case no. 7 in Table 5).

Nevertheless, stormflow response of a particular basin can be modified considerably by manipulation of the vegetative cover. Lal (1983) reported surface runoff from a secondary forest in Nigeria (no. 4 in Table 5) to be negligible. Manual clearing folowed by three years of no-tillage type of annual cropping also produced very little runoff (ca. 1 percent of rainfall) whereas the introduction of conventional tillage after clearing raised the runoff coefficient to about 5 per cent. Mechanical clearing (tractor with shear blade) produced a further increase to about 6.5 per cent, i.e. a sixfold increase compared to the manual case. Since there was no tillage after clearing, a considerable reduction in topsoil infiltrability must have taken place during the latter operation to produce such an increase (cf. Van der Weert 1974; Dias & Nortcliff 1985a,b). The use of tree pushing- and root raking equipment caused still larger increases in surface runoff, viz. ca. 12 percent of incident rain in the no-tillage case (a tenfold increase compared to manual clearing) and ca. 22 percent when followed by convential tillage (a twentyfold increase compared to manual clearing) (Lal 1983).

On the other hand, where hydrological response under natural circum-

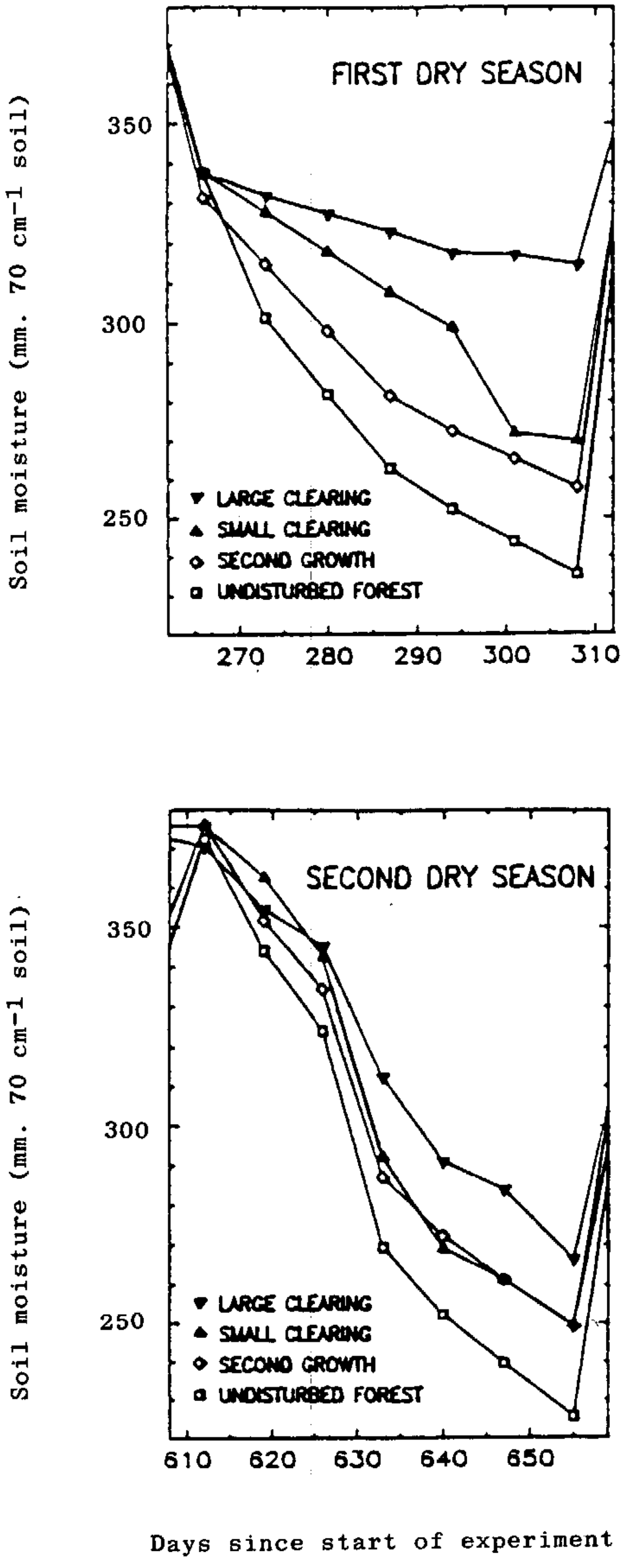

Figure 18 Soil moisture content in undisturbed forest, in a ca. six-year-old secondary forest, and in narrow (10x50 m) and large (50x50 m) clearings in lowland Costa Rica (after Parker 1985).

stances is strongly dominated by SOF (section 2.1), much less dramatic increases in storm runoff may be noted after clearing (no. 1 in Table 5). In such cases, runoff response is already close to what is "physically possible" (Fritsch 1983) and mainly governed by subsoil- rather than top-soil characteristics which are less easily modified (Bonell et al. 1981; Elsenbeer & Cassel 1990). Between the two extremes there is a whole array of hillslope hydrological situations (section 2.2) and, accordingly, also in responses to conversion (Table 5).

As we have seen, replacement of tall vegetation by regenerating scrub, grassland or annual cropping results in reduced ET (Table 4) and therefore higher soil moisture levels (Figure 18). Soil moisture status is an important determinant in basin hydrological response, with wetter conditions corresponding with a more vigorous response and vice versa (Hewlett & Hibbert 1967; Ward 1984).

As such, a certain increase in stormflow volume and peakflow magnitude cannot be avoided after forest conversion, even with minimal surface disturbance (Hsia 1987). Generally, *relative* increases in stormflow after clearing are largest for small events (which may be doubled) and smallest for large events (perhaps in the order of 10 per cent or less; Pearce et al. 1980; Hewlett & Doss 1984). Wherever sufficient care is taken during logging or clearing operations, e.g.in the form of a proper lay-out of the road system, the use of cable logging systems in steep terrain, maintaining a riparian buffer zone and limiting activities to relatively dry conditions, effects can be kept within tolerable limits (Blackie 1972; Pearce et al. 1980; Hewlett & Doss 1984; Subba Rao et al. 1985; Hsia 1987; cf. sections 4.1 and 4.5).

The data collated in Table 5 clearly indicate the strong (local) increases in stormflow volume and peakflow that may be brought about by such adverse practices as regular burning or overgrazing of grassland or forest undergrowth). Similarly, the beneficial effects of reforesting, terracing and contouring degraded lands are equally evident *at this scale* as is the regenerative capacity of most ecosystems when left alone (Patnaik et al. 1974; Gilmour et al. 1987).

Often the clearing of forests in tropical uplands has been seen as the major cause of flooding of large rivers (see Hamilton & Pearce (1987) for

TABLE 5. Effects of (changes in) land cover on stormflow volumes and peakflows: results from tropical catchment experiments#

Study site	Type of land use/ conversion	Effect on stormflow volume	Effect on peakflow
Babinda, Queensland[1]	Selective logging	Not significant	Not significant
	67% clearing	Idem	Small increase at lower end of spectrum
Tekam, Malaysia[2,3]	60% conversion to oil palm	Increase from 19 to 37 % of total flow during 1st three years	38% increase after logging/burning* 65% increase after realignment of stream* 17% decrease after maturation of palms*
	Conversion to cocoa	Reduction from 26 to 21 % of total flow during 1st four years	280% increase*
St. Elie, F. Guyana[4-7]	Clearing for eucalypt	Increases of 410 and 330 mm during 1st two years resp.	
	Clearing for pine	Increases of 495 and 380 mm during 1st two years resp.	
	Conversion to grass on poorly drained soil	Increase of 235 mm yr^{-1} (4 yr)	
	Idem well-drained	Increase of 230 mm yr^{-1} (4 yr)	
Dehradun, India[8]	Secondary scrub to eucalypts	28% reduction in flow	73% reduction during 1st five years
Idem[9,10]	Soil conservation in cultivated basin; forested control	76% reduction in flow over 14 yr (62% in 5 yr)	47% reduction over 1st five years

Idem[10,11]	Cultivation (no soil conservation) vs. grazed forest	Cultivated: 128 mm Forested: 166 mm	Cultivated: 3.5 $m^3 s^{-1} km^{-2}$ Forested: 6.4 $m^3 s^{-1} km^{-2}$
Rajpur, India[12]	20% thinning of grazed forest	Not detectable	9% increase in 1st yr; no effect after 2nd yr
Chandigarh, India[13,14]	Poor scrubland subjected to:	1974 / 1975	1974
	- annual burning	+ 26% / + 218%	+ 225%
	- logging + overgrazing	+178% / + 236%	+ 52%
	- overgrazing	+ 31% / + 552%	+ 47%
	- forestation + trenches	- 75% / - 29%	- 73%
Idem[15]	Poor grass/scrub to forestation, trenches, checkdams, no grazing	60% reduction during 1st six years	61% reduction during 1st six years
Lien Hua-Chi, Taiwan[16]	Clearcut	Not significant	48% increase in median peak discharge
Manankazo, Madagascar[17]	Natural grassland to		
	- burned grassland	Increase of 100 mm yr^{-1}	66% increase in mean annual maximum (11 yr)
	- improved cropping	Reduction of 75 mm yr^{-1}	53% decrease in mean annual maximum (11 yr)
	- pine plantation	Reduction of 80 mm yr^{-1}	69% decrease in mean annual maximum (11 yr)
Kericho, Kenya[18]	54% conversion to tea with soil conservation	High relative but small absolute increases	
	forest vs. built-up area (ca. 13 ha)	Increase from 1% to 36% of rainfall	1140% increase in median peak discharge during 1st year

#for experimental details see footnotes Table 4 and Bruijnzeel & Bremmer (1989) page 91 (Indian studies);
*absolute comparison of one-hour unit hydrographs, no calibration;
[1]Gilmour (1977b); [2,3]DID (1986,1989); [4-7]Fritsch (1983,1987), Fritsch & Sarrailh (1986), J.M. Fritsch, pers. comm.; [8]Mathur et al. (1976); [9,10]Ram Babu et al. (1974), Sastry et al. (1983); [11]Ram Babu & Narayana (1984); [12]Subba Rao et al. (1985); [13,14]Gupta et al. (1974,1975); [15]Kaushal et al. (1975); [16]Hsia (1987); [17]Bailly et al. (1974); [18]Dagg & Pratt (1962)

examples). Although, as pointed out by Hewlett (1982), flood stages of big rivers represent the flow from hundreds of smaller basins, individual stormflow volumes do not simply add up in a downstream direction because of differences in time lag between various tributaries and / or spatial and temporal variations in rainfall and land use. In other words, stormflows generated by heavy rainfall on a cleared part of a large basin may be "diluted" by lesser flows from other parts receiving less or no rainfall at that time. Nevertheless, Hewlett (1982) did not exclude the possibility that widespread forest clearing and subsequent cultivation of uplands in the tropics could have "cumulative effects on valley and river flooding that go beyond the scope of forest management effects on floods".

However, neither Qian (1983) nor Dyhr-Nielsen (1986), using time series of several decades, were able to detect any systematic trends in the magnitude of floods for meso-scale (up to 14500 km^2) basins in Taiwan and northern Thailand respectively which had both undergone extensive "deforestation". They concluded that climatic factors dominated any effects of changes in land use. It should be noted, however, that much of the above "deforestation" probably involved subsequent regeneration rather than widespread dry land cropping (Qian 1983).

Likewise, Mooley & Parthasarathy (1983) did not find a statistically significant trend or oscillation in the occurrence of floods in India between 1871 and 1980 (see also Sakthivadivel & Raghupathy 1978; Jakhade et al. 1984). As discussed in section 4.2 the widespread occurrence of extreme rainfall in India seemed related to large-scale synoptic circumstances. Indeed, truly devastating and widespread flooding is the result of an equally large field of extreme rainfall (Figure 19), especially when the event occurs at the end of the rainy season when soils have become wetted up thoroughly by antecedent rains, leaving relatively little opportunity to accomodate extra water (Hamilton 1987). Under such extreme conditions, basin response will be governed by soil water storage- rather than topsoil infiltration capacities (Bruijnzeel & Bremmer 1989; Gilmour 1977b).

In addition, the extent of flooding in tropical lowlands may be significantly affected by the occurrence of torrential rains in the flood (sic!) plains themselves during times of high ground- and river water levels causing rain water to be "trapped" on the fields (Pal & Bagchi 1975; Rudra 1979; Sen 1979; Raghavendra 1982). Also, backwater effects

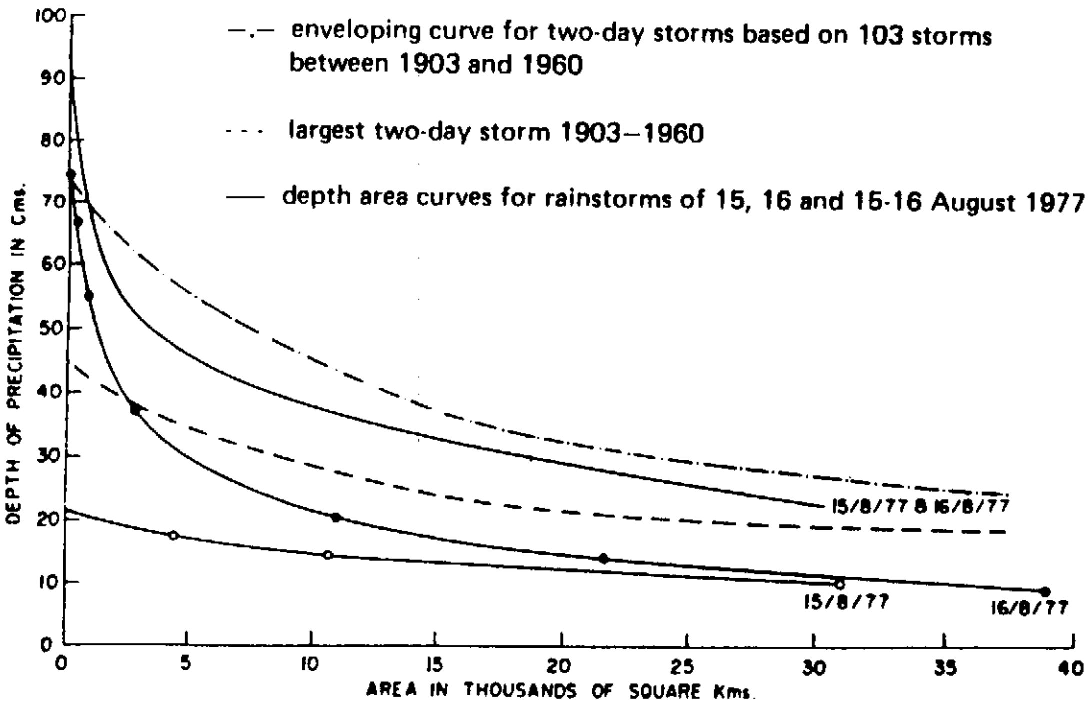

Figure 19. Depth-area curves for extreme rainfall in the Assam valley, north-east India (after Raghavendra 1982).

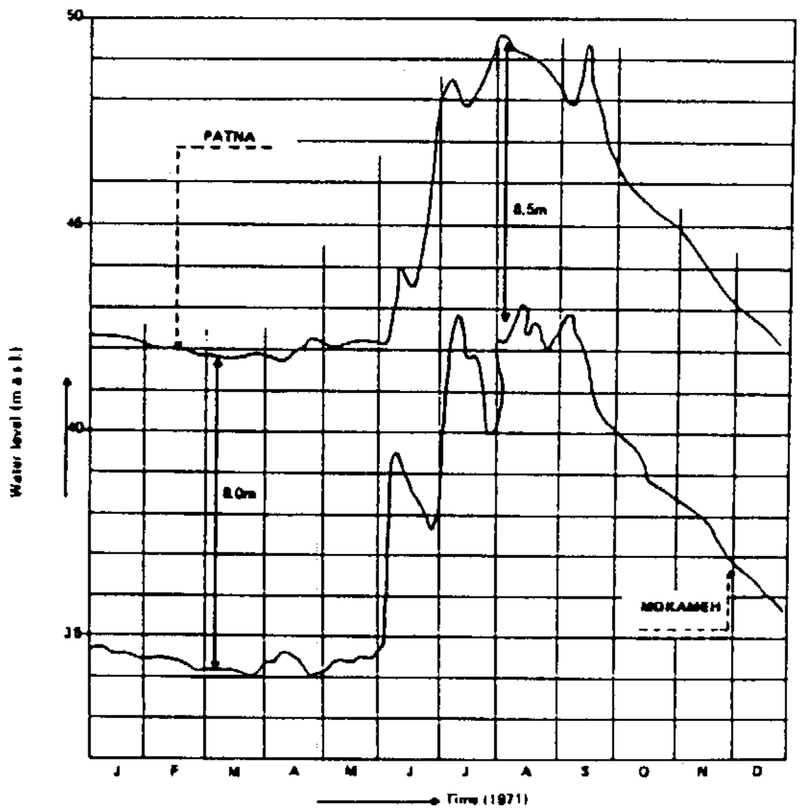

Figure 20. Example of the "back water effect" on the Ganges near Patna (after Dutch Inland Water Transport Mission 1982-1983).

(i.e. the blocking at a river confluence of the flow in one river when the other is in spate) may cause considerable rises in water levels extending for more than 100 km from the confluence (Figure 20) as well as increased sedimentation in the affected river stretch.

Finally, a more indirect effect of extreme rainfall in steep uplands on downstream flooding levels consists of the triggering of large-scale mass movements, especially when earth tremors occur during such wet periods (Goswami 1985). The effect may be twofold: (1) the extra sediment may cause aggradation of the river bed further downstream, thereby increasing the water level in general and flooding hazards in particular (Pal & Bagchi 1975), and (2) the largest of these landslides may temporarily dam a river, producing a devastating surge of water and sediment after the barrier gives way. Such flood waves (as "GLOFS" or glacial lake outburst floods; Galay 1987) may attain heights of 15 to 20 m and travel downstream over hundreds of kilometres (Singh et al. 1974; Mahmood 1987).

Ironically, the construction of barrages to prevent downstream areas from flooding may produce the opposite effect in the middle reaches of meandering rivers whilst at the same time there may be increased bank and bed erosion below the dam as the river tries to regain its lost sediment load, thus causing problems further downstream (Rudra 1979; Mahmood 1987).

One important point still needs to be made. Since it is difficult to assess the size of a flood unambiguously (stage, duration, amount of water or area involved ?), the measure often used for policy making is the economic loss associated with a particular flood. As pointed out by Hamilton (1987), equating economic losses with severity of flooding may introduce serious bias in that it gives the impression that flooding has become more frequent and damaging in recent years, whilst in reality the increased economic losses mainly reflect economic growth and increased floodplain occupancy. Although understandable, it is unfortunate that many (e.g. Eckholm 1976; Bowonder 1982; Murty 1985; Myers 1986) have confused the two ways of expressing severity of flooding.

4.4.2 Dry season flows

In areas where amounts of streamflow available for irrigation or reserv-

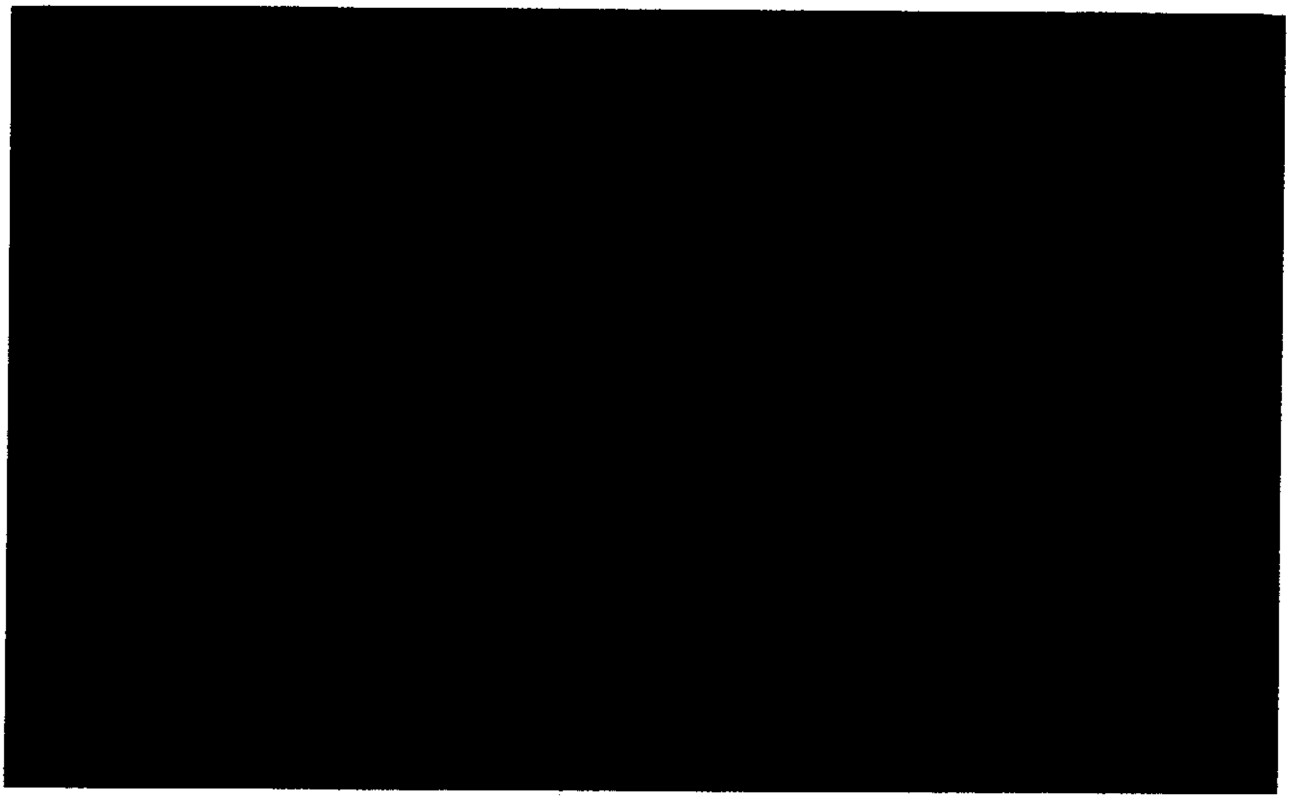

Plate 9. Sediment-laden flash flood resulting from adverse land use practices. Sapi watershed, central Java, Indonesia.

Plate 10.

Reduced dry season flow as a result of diminished infiltration opportunities in the Sapi basin, Java, Indonesia.

Plate 11.

Severe erosion following repeated burning and overgrazing of regenerating scrub, Vanua Levu, Fiji.

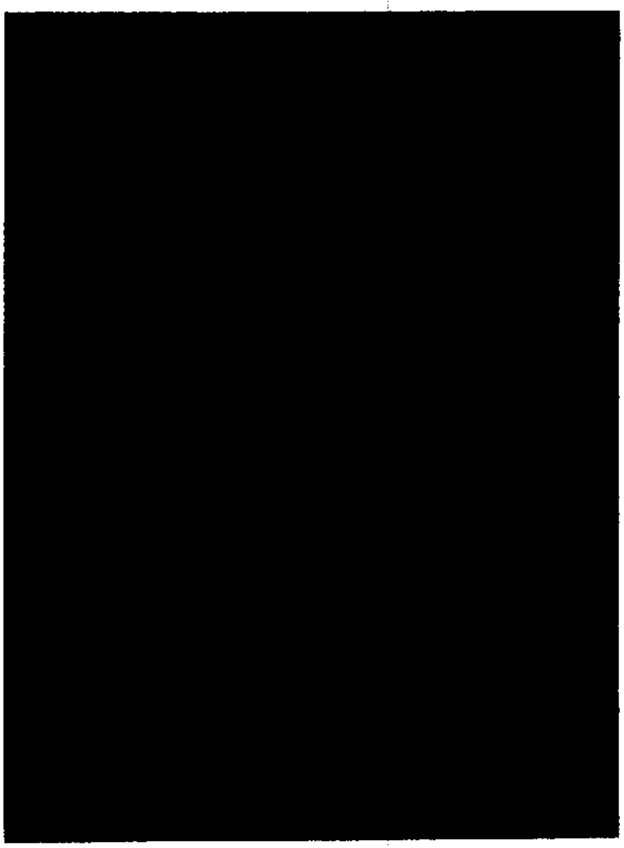

Plate 12.

Deep-seated natural mass movement on forested slope in Central Nepal; note the amount of sediment in various storages such as alluvial fans.

oir storage during the dry season are already critical, any further reductions in flow as a result of changes in land use will assume extra importance.

Reports of greatly diminished dry season flow abound in the literature and are usually ascribed to "deforestation" (Daniel & Kulasingam 1974; Eckholm 1976; Hardjono 1980; RIN 1985; Myers 1986; Nooteboom 1987; Madduma Bandara & Kuruppuarachchi 1988; Bartarya 1989; Figures 21 and 22).

At first sight this would seem to contradict the evidence presented in Table 4 with respect to increases in total water yield following removal of tall vegetation. Also, in most tropical small basin experiments the bulk of this increase in flow was observed during the dry season or baseflow conditions (Gilmour 1977b; Edwards 1979; DID 1986; Abdul Rahim 1989; Figure 23).

However, the conflict can be resolved when taking into account the net effects of changes in ET and infiltration opportunities associated with the respective land use types (Hamilton & King 1983; Bruijnzeel 1989c). Summarising, if infiltration opportunities after forest removal have decreased to the extent that the increase in amounts of water leaving the area as stormflow exceeds the gain in baseflow associated with decreased ET, then diminished dry season flow is the result (Plates 9 and 10).

Reduced infiltration may either result from the use of heavy machinery during forest harvesting or subsequent agriculture (Van der Weert 1974; Lal 1981; Dias & Nortcliff 1985a,b) or from an increase in the area occupied by impervious surfaces such as roads and villages (Ruslan & Manan 1980; Reid & Dunne 1984; Rijsdijk & Bruijnzeel 1990), open cast mining (Bandyopadhyay & Vandana Shiva 1987), overgrazing (Gupta et al. 1974, 1975; Dunne 1979) or improper agricultural practices (Hardjono 1980; Lal 1983). This situation, of course, is widespread in the tropics and can generally be held responsible for the deterioration of streamflow regimes so commonly observed (Figures 21 and 22).

If, on the other hand, surface infiltration characteristics are maintained over most of the area, either because of a well-planned road system and careful extraction (Hsia 1987), a fortunate combination of low rainfall erosivity and stable soil aggreates (Edwards 1979; Figure 23c), or by deliberate soil conservation practices (Table 5; cf. section 4.5), then the effect of reduced ET after clearing will show up as increased baseflow (Figure 23b,c).

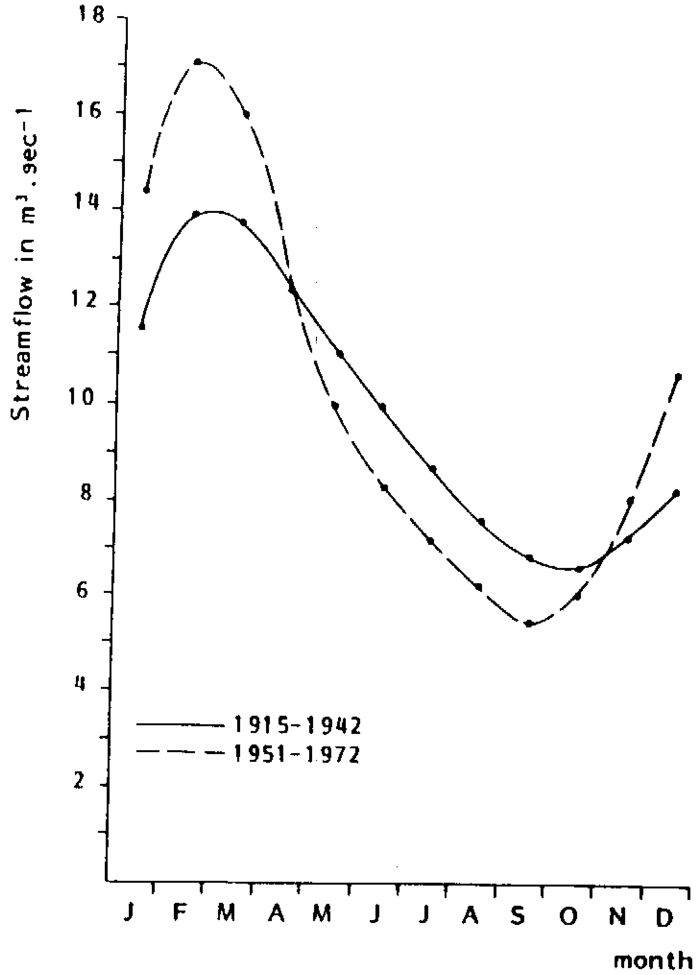

Figure 21. Change in streamflow regime with time in the Konto river basin ($233\ km^2$), East Java, Indonesia (after RIN 1985).

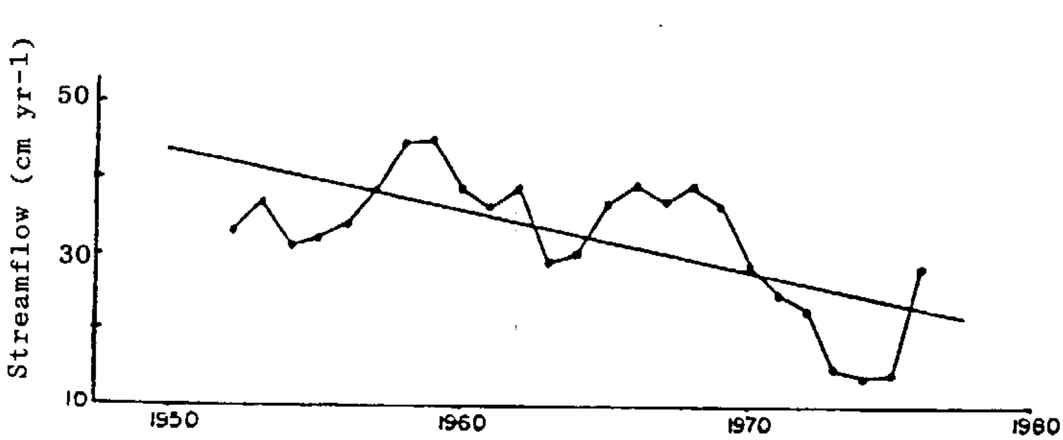

Figure 22. Decline in dry season flow in the Mid-Mahaweli basin, Sri Lanka (after Madduma Bandara & Kuruppuarachchi 1988).

As shown in Figure 23, the effect becomes more prominent as the length of the dry season increases, reflecting the difference in rooting depth between tall vegetation and agricultural crops or grassland (Eeles 1979; Sharda et al. 1988).

It could be argued that reported increases in baseflow have mostly occurred under controlled conditions and as such have limited applicability to the real world situation. Although this is true to some extent, the lesson that can be learned from such experiments is a most important one, namely: the commonly observed deterioration in river regimes following tropical forest removal is not so much the result of the clearing itself but rather reflects a lack of good land husbandry during and after the operation. As pointed out by Bruijnzeel (1986), this is precisely where our hope for the future lies.

The fact noted earlier (Table 4) that total water yield from degraded crop- or grasslands is usually reduced considerably following forestation and even more so after coppicing of the trees, already indicates that the evapotranspirational factor often overrides any gains in infiltrated water. Conditions will differ between sites and any analysis should take into account prevailing rainfall intensities, infiltration capacities before and after tree planting, hillslope hydrological patterns (occurrence of SOF, etc.) as well as differences in rainfall interception and transpiration between the two vegetative covers (Bruijnzeel & Bremmer 1989; Institute of Hydrology 1990).

It is unfortunate that the only study dealing with the hydrological effects of reforesting severely eroded tropical agricultural uplands that I am aware of (Hardjono 1980; Figure 24), is not very stringent (see footnotes Table 4 for experimental details).

Accepting Hardjono's data at face value, they could be taken as evidence that reforesting degraded land restores dry season flow (inset of Figure 24). This would imply that the higher ET of the forest would be more than compensated for by increased infiltration following forestation. However, it is difficult to reconcile this with the fact that total water yield from the (354 ha) forested basin was some 580 mm less than that for the (207 ha) agricultural catchment (Table 4; cf. Edwards 1979). Also, in view of the limited moisture storage capacities of the soils in

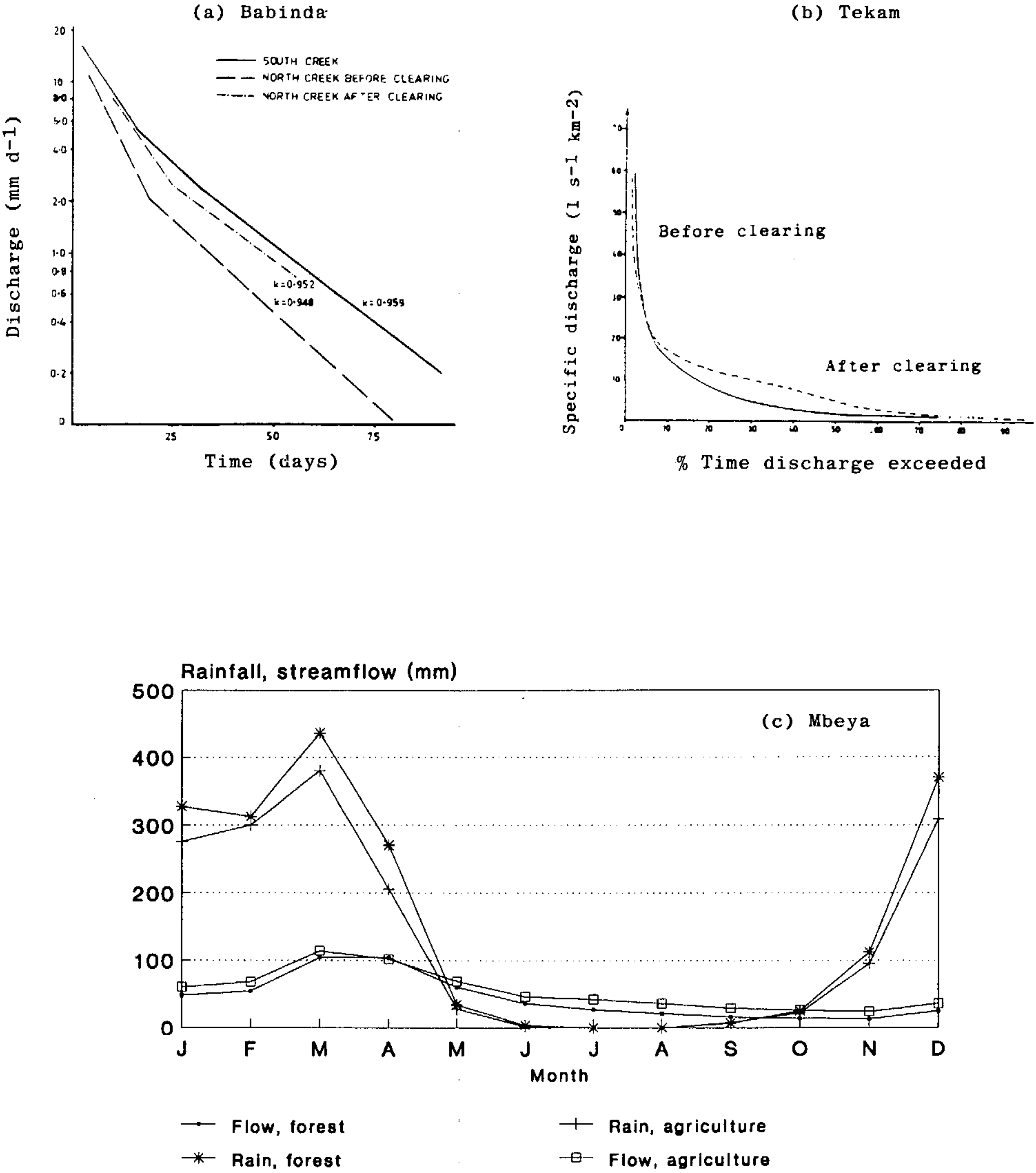

Figure 23. Comparison of dry season flows before and after forest clearance. (a) Babinda, Queensland (after Gilmour 1977b), (b) Sungei Tekam, Malaysia (DID 1986), (c) Mbeya, Tanzania (Edwards 1979); for site and experimental details see footnotes Table 4.

h catchments (as indicated by the rapid drop in flow during periods of y rainfall) and the severe depletion of moisture reserves at the end of dry season in the forested basin (slow return of flow in November and ember), it would seem more likely that the higher "summer" flows in the ested catchment reflect its larger (by 71 per cent) basin area rather n anything else.

Clearly, more (rigorous) work is required to arrive at a firm answer to question whether reforestation of shallow degraded soils will improve season flows or not (cf. chapter 6).

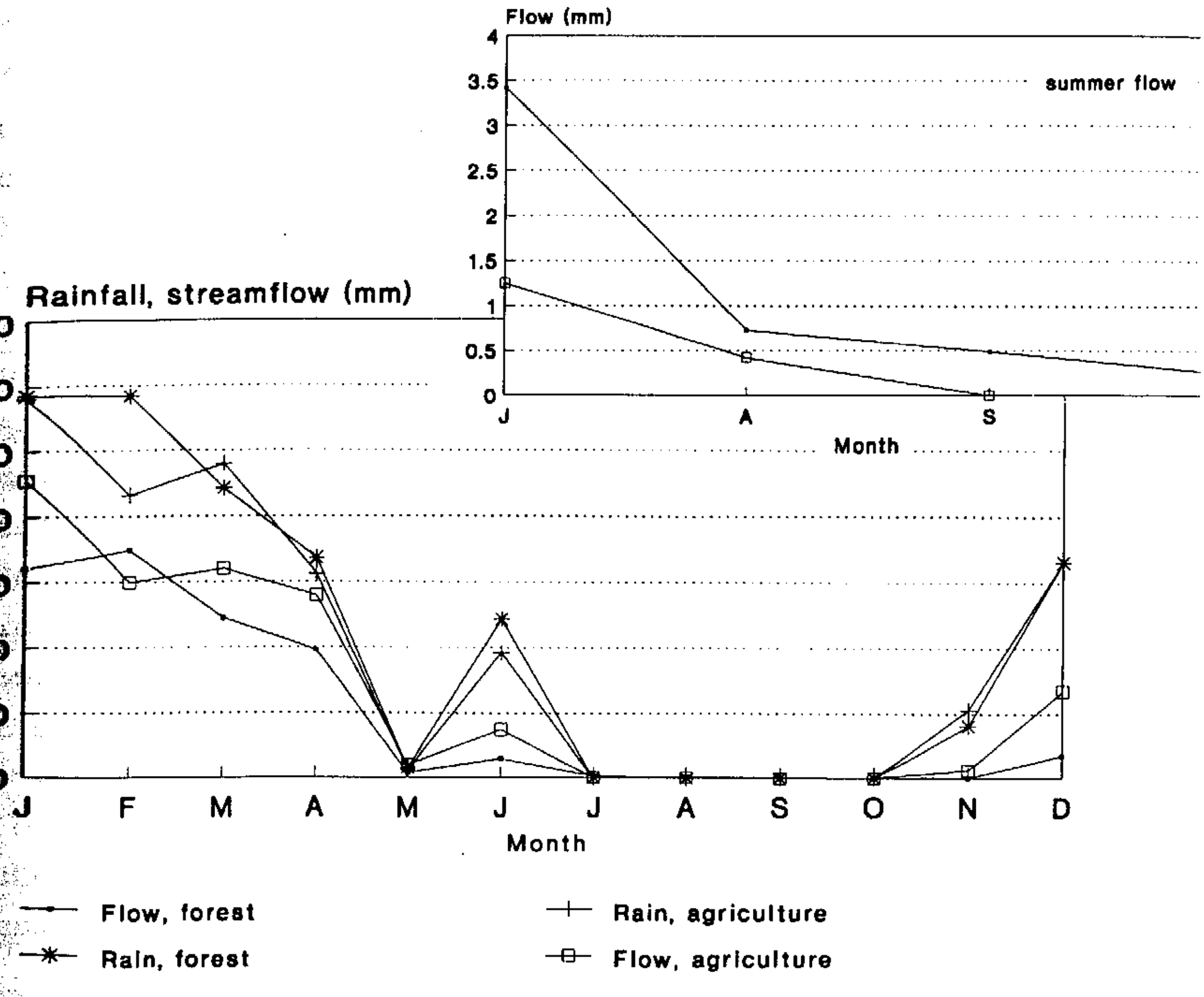

Figure 24. Monthly rainfall and streamflow for forested and agricultural basins at Pidekso, central Java (based on data presented by Hardjono 1980).

4.5 Effect on sediment production and yield

In contrast to the misconceptions regarding the influence of forests on rainfall, water yield and major floods discussed in the previous sections, the popular view of trees (particularly when planted in dense stands) as checkers of soil erosion by and large is supported by the scientific facts.

When dealing with the effects of changes in land use on erosion and sedimentation, it is helpful to distinguish between surface erosion (splash, sheet and rill erosion), gully erosion, and several forms of mass movements, since the ability of a vegetation cover to control the various forms of erosion is rather different (Plates 11, 12 and 14).

It is equally important to make a distinction between on-site erosion (i.e. on the scale of a field or a hillslope) and off-site / downstream effects. Often only part of the material eroded from a hillside will enter the drainage network, the rest may move into temporary storages in depressions, footslopes, small alluvial fans (Plate 12), or be deposited in the beds of ephemeral tributary drainages, behind debris or on flood plains. This stored material may be transported again during large storms or become colonised by vegetation and form a stable topographic element for decades (Dietrich & Dunne 1978; Trimble 1981).

As the number of storage opportunities tends to increase with catchment size, the ratio between on-site erosion and amounts of sediment carried by a stream (the "sediment delivery ratio ratio") decreases markedly with basin size (Walling 1983). Naturally, effects of erosion will be felt much more quickly on-site than further downstream. For example, soil losses from agricultural fields may lead to such a decrease in productivity that a farmer is forced to abandon them (Shrestha 1988), although this sediment may hardly show up in the streams of the area. On the other hand, as will be shown below, it may take decades before any reductions in surface erosion in upland areas are reflected in reduced sedimentation downstream in the floodplains of large rivers (Goswami 1985; Pearce 1986).

In the following, we will discuss the effects of changes in vegetation and land use patterns on rates of surface- and gully erosion (section 4.5.1), on mass wasting (section 4.5.2), and on stream sediment loads (section 4.5.3) in the humid tropics.

4.5.1 Surface and gully erosion

Wiersum (1984) has reviewed results from about 80 studies of surface erosion in (sub)tropical forest and tree crop systems (Table 6).

Table 6. Surface erosion in tropical forest and tree crop systems ($t\ ha^{-1}\ yr^{-1}$; Wiersum 1984)

	Min	Median	Max
1. Natural forests (18/27)*	0.03	0.3	6.2
2. Shifting cultivation, fallow period (6/14)	0.05	0.2	7.4
3. Plantations (14/20)	0.02	0.6	6.2
4. Multi-storied tree gardens (4/4) (4/4)	0.01	0.1	0.15
5. Tree crops with cover crop/mulch (9/17)	0.10	0.8	5.6
6. Shifting cultivation, cropping (7/22)	0.4	2.8	70
7. Agricultural intercropping in young forest plantations (2/6)	0.6	5.2	17.4
8. Tree crops, clean-weeded (10/17)	1.2	48	183
9. Forest plantations, litter removed or burned (7/7)	5.9	53	105

* (a/b) a = number of locations
b = number of "treatments"

Although the data collated in Table 6 are of variable quality and reflect a variety of pedological situations, they clearly show erosion to be minimal in those ecosystems where the soil surface is adequately protected by a well-developed litter- and herb layer (no's 1-4). Erosion rates may increase only slightly upon removal of the understorey (no. 5), but they rise dramatically when the litter layer is destroyed or removed (no's 7-

9). The initial effect is rather small due to the effect of residual organic matter on soil aggregate stability and infiltration capacity (no's 6 and 7; Coster 1938; Gonggrijp 1941a; Wiersum 1985), but repeated disturbances, such as burning or frequent weeding, have much more serious consequences (no's 8 and 9 in Table 6; cf. Bailly et al. 1974; see Wiersum (1984) for details). Bruijnzeel & Bremmer (1989) collated a similar set of data for the Ganges-Brahmaputra basin and arrived at the same conclusions.

Incidently, the above data suggest that the protective value of tree stands lies not so much in the ability of the tree canopy to break the power of raindrops, but rather in developing and maintaining a litter layer (Dalal et al. 1961; Wiersum 1985). Indeed, several recent studies have shown that the erosive power of rain dripping from forest canopies in various tropical and warm-temperate parts of the world may be substantially larger than for rainfall in the open, reflecting the larger drop size of canopy drip (Mosley 1982b; Wiersum 1985; Vis 1986; Brandt 1988). As long as the complex of litter, herbs and understorey remains relatively undisturbed, it is able to deal with this increased striking force quite effectively, but, as already indicated, its removal may create problems if not substituted by soil conservation practices (Table 6).

High rates of surface runoff and erosion have often been reported for the second year of the cropping phase of shifting cultivation (Toky & Ramakrishnan 1981; Hurni 1982; Mishra & Ramakrishnan 1983; Sato et al. 1984; Das & Maharjan 1988) and for heavily grazed and/or annually burned grassland areas (Jasmin 1975; Costales 1979; Impat 1981). Smiet (1987) has made an important point in this respect. He drew attention to the fact that the margins for forest management re soil protection are much broader than those associated with non-forest types of land use, notably grassland or cropping. The degraded natural and plantation forests encountered in many tropical uplands are still able to fulfil their protective role since gaps are generally rapidly colonised by pioneer species (Coster 1938; Rijsdijk & Bruijnzeel 1990). Conversely, the margins associated with grazing or agroforestry are much more easily exceeded (fire, overgrazing, shallow landslides, etc.; Smiet 1987).

On the other hand, surface erosion from well-kept grassland, moderately grazed forests and agricultural fields with appropriate soil conservation measures (alley cropping, bunding or terracing, mulching, no-tilllage or minimum tillage, etc.) are generally low to moderate (Roose 1977; Khybri et al. 1978; Mensah-Bonsu & Obeng 1979; Impat 1981; Lal 1983,1990; cf. Hudson 1971).

Although it has been realised for a long time that the construction and use of logging tracks and roads constitutes the main cause of increased basin sediment yields following logging operations in temperate and tropical forests (Megahan & Kidd 1972; Gilmour 1971; Rothwell 1978), comparatively little information is available on amounts of sediment generated on "regular" trails and roads, villages and mining spoils in the tropics, even though these are widely acknowledged as important sources of runoff and sediment (Van der Meer 1981; Hamilton & King 1983; RIN 1985).

Henderson & Rouysungnern (1984) presented a double mass curve of rainfall versus annual basin sediment yield for the 1754 km^2 Mae Taeng basin, northern Thailand, which exhibited two striking breaks in slope which they ascribed to incremental road construction. They also quoted work by Kasem Chunkao and co-workers reporting considerably enhanced sediment yields for basins draining tin mines as well as erosion rates for a village settlement in the northern hills that were "36 times those from evergreen forest".

Evidence of dramatically accelerated sediment production associated with road construction in a well-vegetated catchment of 112 ha in eastern Thailand was presented by Henderson & Witthawatchutikul (1984). They estimated a total sediment load of about 105 t ha^{-1} during the first year of road usage, whilst before construction began, the water of the (intermittent) stream had been potable even during rain storms, suggesting very low outputs of sediment under undisturbed conditions (cf. Greer et al. 1989). Soon after the onset of the rains, gullies started to form in the loose roadside spoil, especially at culvert outlets, and extended rapidly during the first few months. After that, the main contributors of sediment were slumps and landslides in the cutbank of the road (Henderson & Witthawatchutikul 1984; see also section 4.5.2 on mass wasting).

More recently, data on amounts of sediment produced by foot paths in areas with "taungya" cultivation (agricultural intercropping in young forest plantations) in upland Java were reported by Bons (1990) and Rijsdijk & Bruijnzeel (1990). Both studies found surface erosion from the recently cleared fields themselves (as determined by Wischmeier-type plots) to be negligible, but on the basis of measurements of sediment outputs from hillslope micro-catchments compacted field boundaries were shown to supply considerable amounts of sediment, viz. ca. 34 and 70 t ha^{-1} yr^{-1} respectively. However, a gradual deterioration of surface conditions after clearing often occurs as organic matter disappears and the soil is exposed to rain, wind and sun (El-Swaify et al. 1982; Lal 1987) and typical erosion rates for terraced dry land cropping (maize) of volcanic soils that were cleared decades ago in East Java range from 10 to 50 t ha^{-1} yr^{-1} (although much higher losses (up to 500 t per 3 mo) were observed on fields with onions planted in rows up and down the slope (Rijsdijk & Bruijnzeel 1990)).

Very high soil losses were also observed for two large unbounded plots in the same area in rural settlements (0.16-0.27 ha), viz. ca. 170 and 215 t ha^{-1} yr^{-1}, whereas corresponding mean annual runoff coefficients amounted to 26 and 40 per cent of rainfall respectively (Rijsdijk & Bruijnzeel 1990). The same authors also presented sediment losses from cobbled road surfaces in reasonable condition (typically about 25 t ha^{-1} yr^{-1}) and from unpaved roads in poor condition (ca. 70 t ha^{-1} yr^{-1}) which exhibited mean runoff coefficients of ca. 35 and 65 per cent respectively. Roadside ditches draining road surfaces and adjacent agricultural fields and yards typically carried about 10 t ha^{-1} yr^{-1}, suggesting that a fair proportion of on-site eroded material did not reach the drainage network and hence the streams (Rijsdijk & Bruijnzeel 1990; cf. section 4.5.3).

Small rills started to develop in the above-mentioned compacted field boundaries during the second year of "taungya" cultivation (Bons 1990) and remained a source of runoff after the area was covered in scrub (surrounding the young trees) again (L.A. Bruijnzeel, personal observation). Indeed, when such incipient gullies are not treated at an early stage, they may soon reach a point where lateral and headward extensions through scouring, undercutting and subsequent collapse of the walls have become so intense that their restoration becomes a difficult and costly affair (Hudson 1971; Haigh 1984b; Morgan 1986).

Bergsma (1977) and Rijsdijk & Bruijnzeel (1990) could trace the occurrence of active gullying in parts of Central and East Java to improper discharging of surface runoff from agricultural fields. Similarly, in the

Himalaya and other tropical areas experiencing a distinct dry season, the clearance of vegetation and subsequent overgrazing of land with relatively erodible soils (such as those often found on sand- and siltstones, or quartzites, gneisses or lacustrine deposits), have led to intense gullying (Brunsden et al. 1981; Haigh 1984b; cf. Plate 11).

However, gullying may also be initiated in undisturbed rain forested terrain, e.g. when soil becomes exposed through treefalls or landslips (Ruxton 1967; Turvey 1974) or during extreme rainfall (Herwitz 1986a), whereas in some cases gullies may be formed by the collapse of subsurface pipes (Morgan 1986).

Once initiated, the influence of vegetation on actively eroding gullies is rather limited and rehabilitation schemes employing vegetative means will often need to be supplemented with mechanical measures, such as check dams, retaining walls as well as protected waterways diverting the water from the eroding headwall (Hudson 1971; Blaisdell 1981; Narayana & Sastry 1985). As indicated already, restoring gullies is difficult and expensive. As such, to avoid their occurrence through sound land husbandry is obviously much better than having to rehabilitate them.

4.5.2 Mass wasting

Some of the highest reported natural erosion rates from (rain) forested areas (Pain & Bowler 1973; Li 1976) were related to intense mass wasting under conditions of steep topography, tectonic activity and intense rainfall. Most authors ascribe the high intensity of mass movements in tectonically active tropical steeplands to a combination of geological and climatic rather than land use factors. Steep dipslopes, unstable nature of rocks due to their structural disposition (e.g. degree of fracturing), depth and degree of weathering, high seismicity in certain areas (e.g. the Pacific rim and the Himalaya), and oversteepening of slopes through undercutting by rivers, rank among the most important geological factors (see reviews by Ramsay (1986,1987b) and Whitehouse (1987)).

Of particular interest is a study by Prasad (1975), who discussed ten years of observations of seismic activity, rainfall and landslide occurrence in part of the Kosi basin in eastern Nepal. In general, landslides were most frequent during times of both rainfall and earthquake activity (mainly in July and August; cf. Pain & Bowler 1973). Since slides also occurred during times of low seismicity, Prasad (1975) concluded that intense precipitation and the associated saturation of soils were apparently more important than seismic shocks.

The latter contention was supported by many other observations in the region. For example, Carson (1985) related how in one area in the Middle Hills of Nepal, villagers indicated that the landslides still visible in the early 1980's had all occurred during two events of heavy rain, one in 1934 (!) and the other in 1971 (cf. Manandhar & Khanal 1988). Similarly, Starkel (1972) found that during an extreme rainfall event of more than 700 mm in three days near Darjeeling, north-eastern India, many new landslides were initiated and old ones reactivated. He estimated the associated erosion rate at about ten times the annual average.

With such strong geological and climatic controls over mass movement processes in steep terrain, it is rather difficult to evaluate the influence of various disturbances on land-slide frequency and magnitude with any degree of certainty. Also, remaining areas of forest in the tropics are often found on slopes too steep for terraced cultivation. This immediately introduces the methodological difficulty of finding comparable control sites (the forested slopes being steeper and therefore more susceptible to gravity). In addition, it is not uncommon that such breaks in slope reflect a change in lithology as well (Kienholz et al. 1983,1984).

As for the influence of (tall) vegetation on slope stability, the net effect is generally considered positive, the major factor being mechanical reinforcement of the soil by the tree roots (Ziemer 1981; O'Loughlin 1984). Although the removal of tall vegetation may lead to wetter conditions in the soil due to reduced ET (section 4.3), which would tend to increase slide hazard, this is usually not thought to be very important. After all, often most failures occur during the second half of the rainy season (Prasad 1975; Carson 1985; Rijsdijk & Bruijnzeel 1990), when soils will have become thoroughly wetted by antecedent rains anyway. Under such conditions, the extra cohesion imparted by tree roots may be critical to slope stability.

It is important, however, to make the distinction between deep-seated and shallow (less than, say, 3 m) slides, as the former do not seem to be influenced appreciably by the presence or absence of a well-developed root network (Starkel 1972; Carson 1985). For example, Brunsden et al.

(1981) reported how mass wasting in phyllitic terrain in eastern Nepal during a few heavy storms in late July 1974, was much more intensive on steep forested slopes than in more gently sloping cultivated areas. Failures were generally restricted to deep ravine headwater areas and along the lower valley sides and banks where undercutting occurred.

The contention of Starkel (1972), that the role of vegetation in preventing shallow slope failures (often triggered during heavy rain) is "most important", was demonstrated rather dramatically by the work of Manandhar & Khanal (1988) in an area underlain by limestones and phyllites, some 20 km south of Kathmandu, Nepal. Examination of aerial photographs taken in 1972 and 1986 showed an increase in the number of landslide scars from 93 to 743. Most of these failures occurred on slopes steeper than 33° and had been triggered during a single cloudburst in September 1981. Only a few landslides had occurred in the thickly vegetated headwater area of the catchment, the majority being found near limestone quarries and on sparsely vegetated slopes (Manandhar & Khanal 1988; cf. Haigh 1982,1984a).

Although numerous, these small and shallow failures found in mid- or upper slope positions usually heal rather quickly (Ramsay 1986; Euphrat 1987). In addition, they are generally only modest contributors of sediment to streams as they become rarely fully incorporated in the drainage network (Ramsay 1987a; Rijsdijk & Bruijnzeel 1990; cf. section 4.5.3), in contrast to deeper forms of mass wasting (Brunsden et al. 1981; Ramsay 1987a; cf. Plate 12).

Ramsay (1986) distinguished two categories of disturbance: (1) changes in land use (principally the removal of forest cover, followed by grazing or cultivation, possibly with terracing and irrigation) and (2) construction activities (mainly roads, irrigation canals and housing).

Terracing of hillslopes after vegetation removal was generally not considered a direct cause of mass wasting although Marston (1989) noted that poor control of terrace drainage in parts of Nepal was important in this respect (cf. the observations of Bergsma (1977) and of Rijsdijk & Bruijnzeel (1990) in Java). As pointed out by Carson (1985), the length and intensity of human occupation is often such, that areas liable to sliding due to addition of irrigation water would probably have done so a long time ago (cf. Plate 15).

Rather, existing irrigated terraces are stable and small slumps and collapsed terrace risers (Euphrat 1987) quickly repaired. As described by Johnson et al. (1982), Himalayan hill farmers are well aware of increased slide hazards associated with the accumulation of water on terraces. This perception has sometimes led them to shift from irrigated to rain-fed cropping. A consequence of this practice, however, is an increase in surface erosion, as rainfed terraces are often deliberately outward-sloping in order to dispose of excess rainfall during the monsoon, which could damage the crops by waterlogging (Johnson et al. 1982). A similar observation was reported by Rijsdijk & Bruijnzeel (1990) for onion cultivation in Indonesia.

According to Ramsay (1986), irrigation canals in upland areas are frequently associated with slope failures due to both the removal of toe support from slopes and to saturation of the weathering mantle by seepage and overflow. This brings us to the effects of *construction activities* on mass movements and sediment production.

The construction of large dams and subsequent inundations of valleys will have on-site and off-site consequences. Around the reservoir itself, the increased pore pressure associated with the saturation of the slopes may well trigger slides when the water level in the lake is lowered for some reason whereas below the dam, a new cycle of riparian mass wasting may be initiated as the river will tend to regain its lost (i.e. trapped by the dam) sediment load by increased incision (Rudra 1979; Carson 1985; Mahmood 1987; Galay 1987).

However, by far the most important construction impact on slope stability in many tropical steeplands, is the building of roads (Haigh 1984a; Henderson & Witthawatchutikul 1984; Henderson & Rouysungnern 1984; Rijsdijk & Bruijnzeel 1990). Although it can be shown that proper road engineering can solve many of the problems (Schaffner 1987; Adams & Andrus 1990), it should be realised that associated costs are extremely high, especially in mountainous terrain, both during construction (e.g. up to more than one million dollars km^{-1}; Carson 1985) and afterwards (i.e. maintenance; cf. Adams & Andrus (1990) for various suggestions on how to reduce such costs).

The next section will examine to what extent changes in on-site erosion, both in the positive and negative sense, will show up in basin sediment yields.

4.5.3 Basin sediment yield

As indicated before on several occasions, it is rather hazardous to compare water yields for catchments with different land uses and simply ascribe differences in yield to the contrast in vegetation (section 4.3). This is even more true in the case of basin sediment yields, as will be shown below.

Not only may a single extreme event change the entire sediment picture of an area overnight by releasing enormous amounts of sediment at once which may then move into various types of temporary storage influencing yields for quite some time (Starkel 1972; Pain & Bowler 1973; Trimble 1981; Goswami 1985), but also sediment yield figures will depend strongly on sampling intensity and computation methods (Dunne 1977; Walling 1977). In addition, there is an effect of basin size (i.e. number of storage opportunities; Walling 1983).

It is well-known that often the bulk of annual sediment transport occurs during a limited number of high rainfall events (Douglas 1967a; Turvey 1974; Sarma 1986; Amphlett 1988; White 1990; Figure 25). As a result, it is imperative that these high flows be sampled adequately or else a gross underestimate of sediment yield will be obtained. A rather dramatic example of this can be found in the work of Biksham & Subramanian (1988) on the Godavari river in Central India. Based on occasional sampling during the various seasons over a period of three years, the authors computed a mean suspended sediment concentration of 770 mg l^{-1}. The corresponding value based on a year's sampling on a daily basis by the Central Water Commission amounted to 1525 mg l^{-1}, whilst the 10-year average concentration (daily observations) was 1845 mg l^{-1} (Biksham & Subramanian 1988). In this case, the annual suspended sediment load would have been underestimated by as much as 240 per cent. A similar case was reported by Kaatee (1989) for a 1235 ha basin in East Java.

As for the lingering influence of extreme events, one cannot do better than refer to the work of Goswami (1985) who reported on the effects of the August 1950 earthquake in Assam on the sediment load of the Brahmaputra river in the Assam valley between 1955 and 1979. During this earthquake, apparently one of the most severe ever recorded, massive landsliding occurred which temporarily blocked several major tributaries draining steep forested terrain. Bursting of these dams after several days (!) not

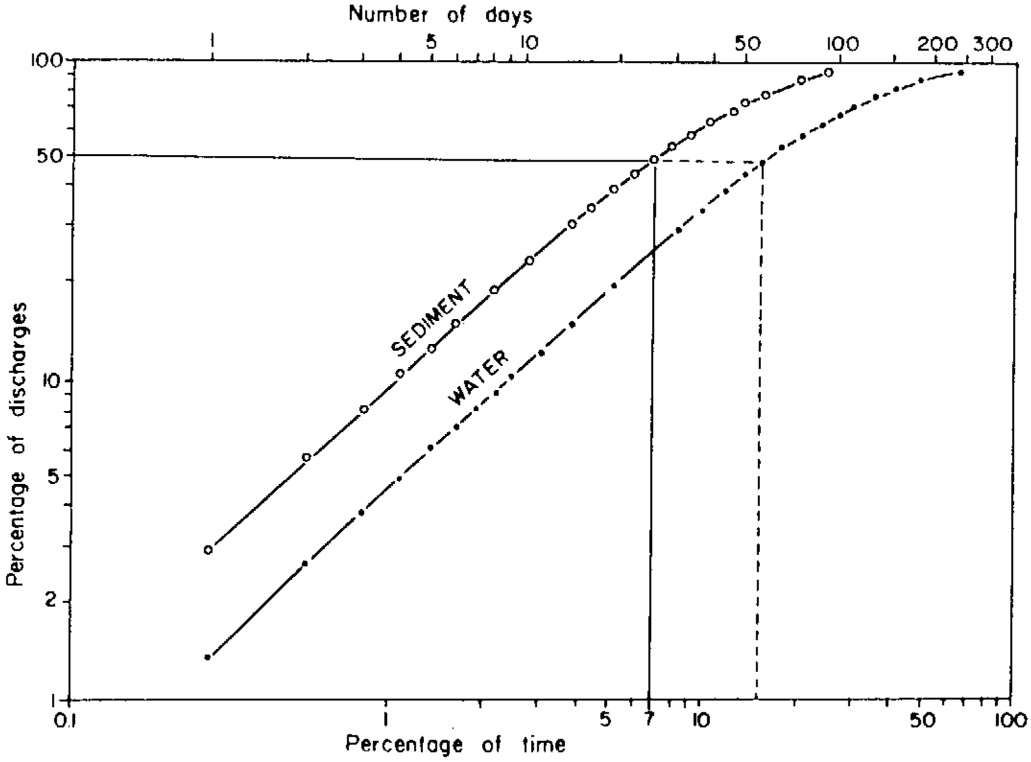

Figure 25. Cumulative percentages of suspended sediment- and water discharges on the Burhi Dihing river, Assam, against cumulative percentage of time for the year 1974 (after Sarma 1986).

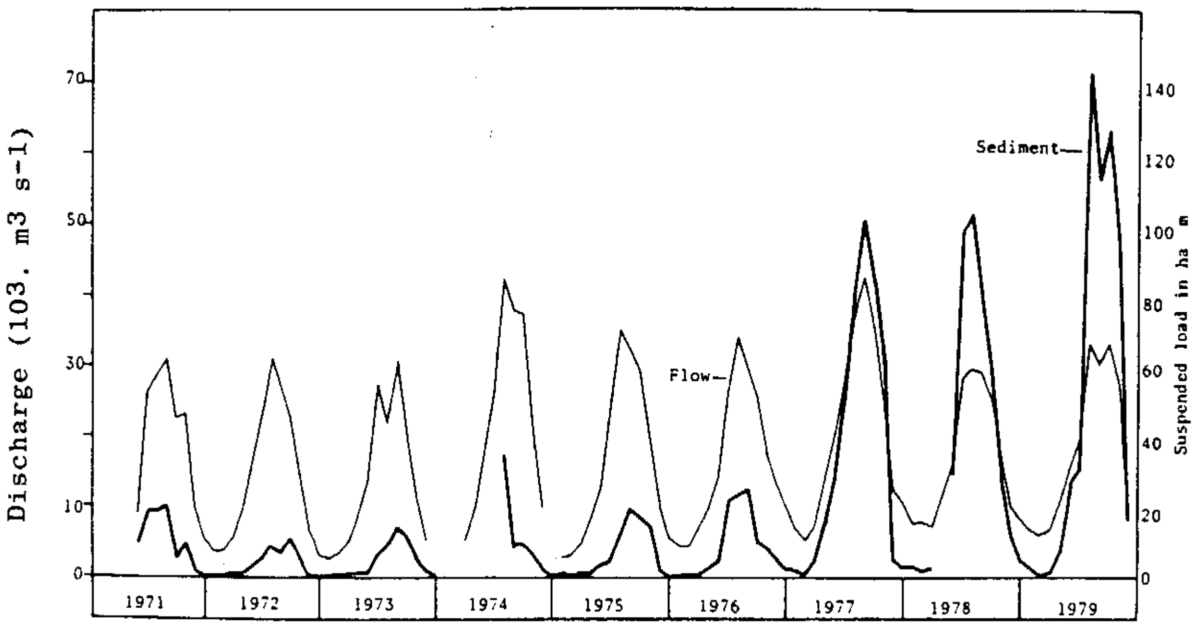

Figure 26. Mean monthly flow and sediment discharge for the Brahmaputra river at Pandu, Assam, 1971-1979 (after Goswami 1985).

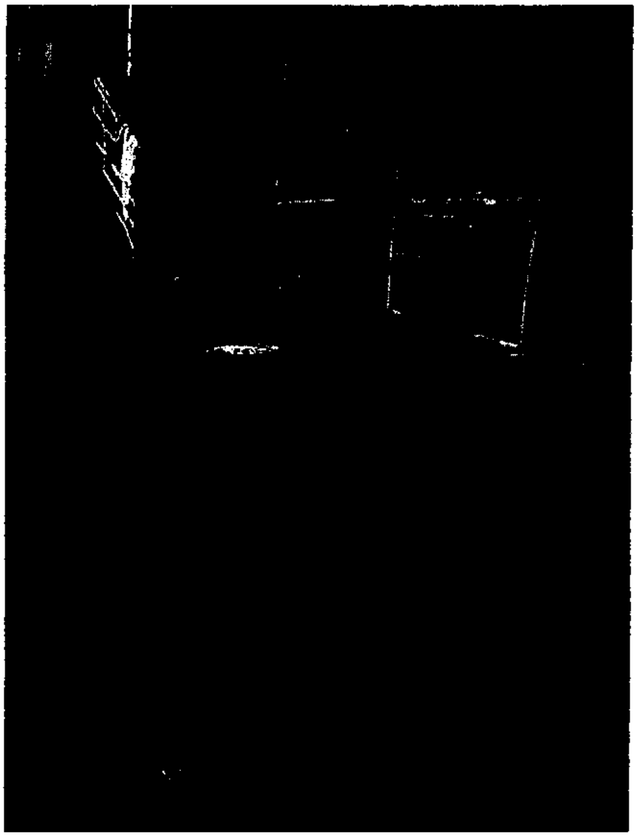

Plate 13.

Sediment production from settlements may be considerable but is often overlooked; Tulungrejo village, East Java, Indonesia.

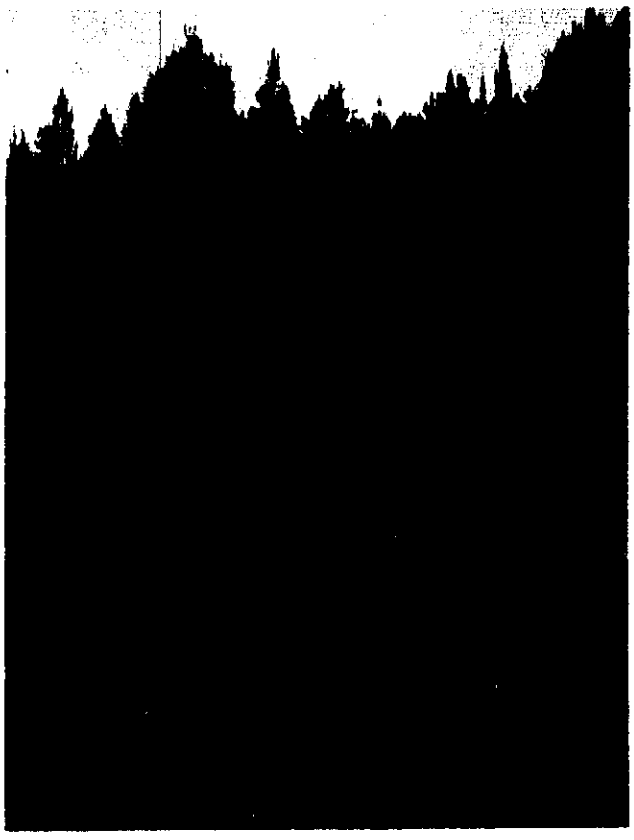

Plate 14.

Forestation of degraded land may check most surface erosion but may not be enough to stop gully erosion; also, there may be problems of reduced dry season flows and long-term productivity; Vanua Levu, Fiji.

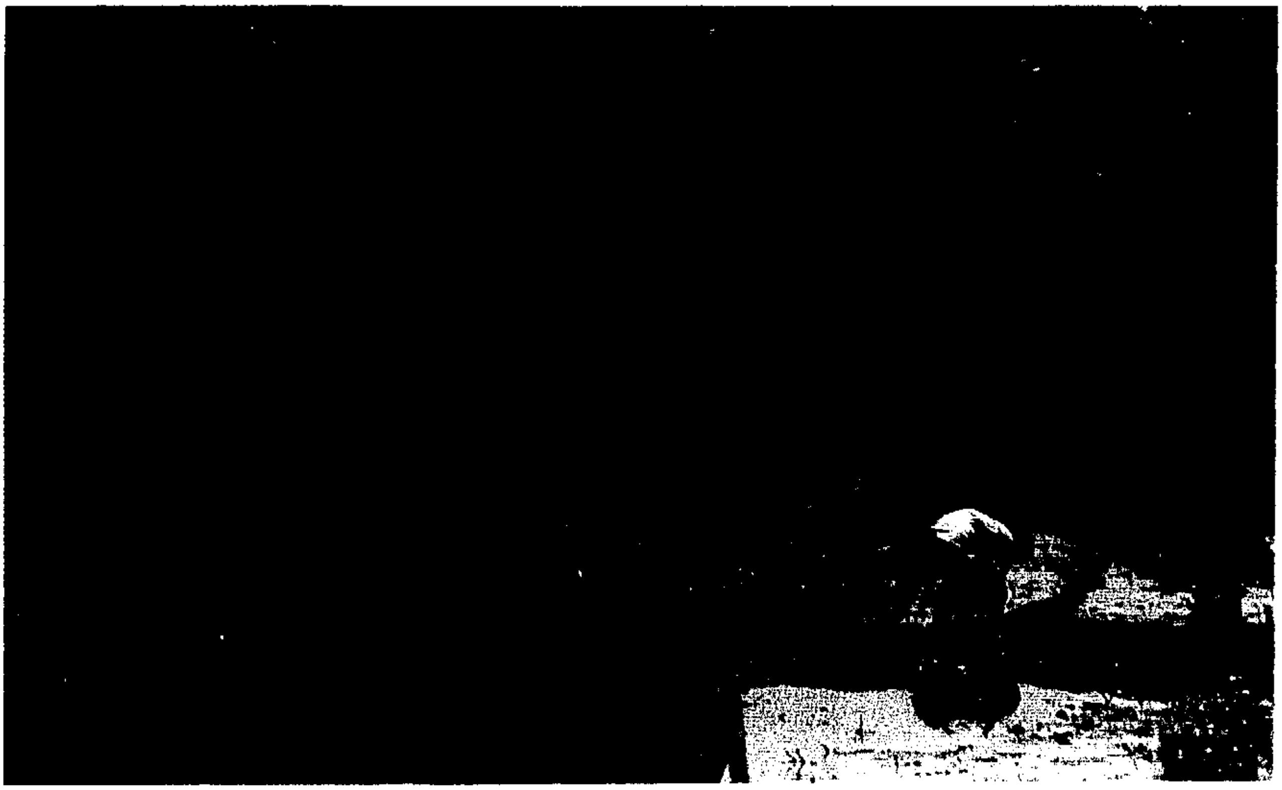

Plate 15. The cultivation of rice on irrigated terraces (Central Java) is a good example of sustainable agriculture in the humid tropics (photograph by Rob Mieremet).

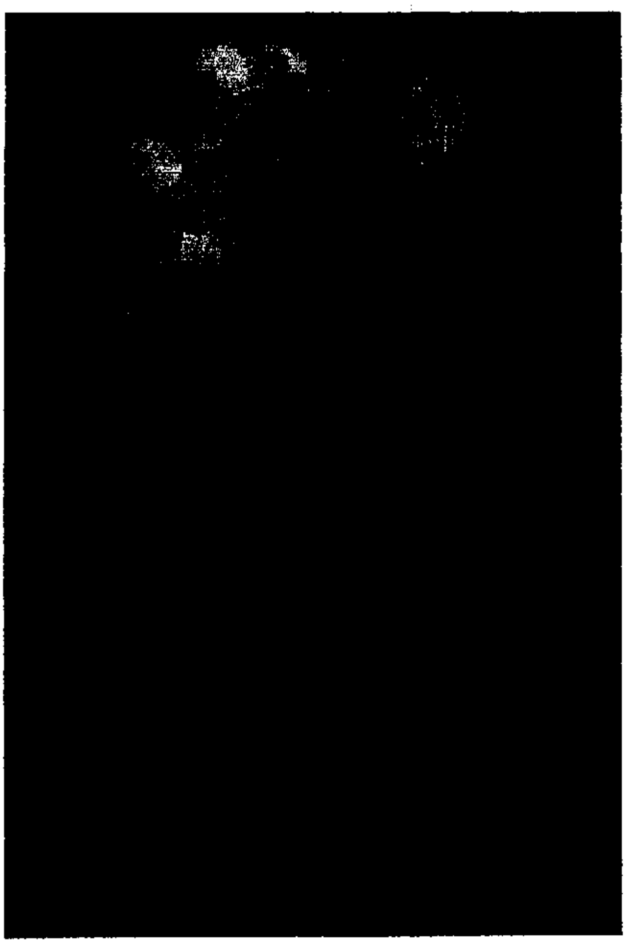

Plate 16.

Certain forms of agroforestry such as this home garden cum field in Java have excellent soil conserving properties.

only produced devastating floods downstream, but also brought down enormous volumes of sediment, thereby raising the beds of the above rivers considerably (Poddar 1952). The mean annual suspended sediment load and water discharges of the Brahmaputra at Pandu (600 km downstream) between 1955 and 1963 amounted to 750,000 m^3 and 16,530 m^3 s^{-1} as against 130,000 m^3 and 14,850 m^3 s^{-1} from 1969 to 1976, respectively (Goswami 1985). During the former period the river reach upstream of Pandu was aggrading, whilst the channel was degrading between 1969 and 1976.

However, trends in ag- or degradation at a particular channel cross section may not be indicative for the entire river reach. For example, the increased sediment discharges observed in the late 1970s at Pandu (Figure 26) appeared to reflect temporary degradation of the reach immediately upstream of the gauging site whilst that immediately downstream experienced significant aggradation during that time (Goswami 1985). An analysis of aggradation and degradation rates for the Brahmaputra river bed over a stretch of more than 600 km and over several decades revealed the following (Goswami 1985):

- There was a considerable gain in sediment in all reaches except one, with aggradation ranging from 0.5 to 2.4 (!) m between 1957 and 1971 (i.e. up to 20 years after the earthquake);

- Between 1971 and 1977 an average degradation of about 20 cm was determined, i.e. only a small fraction of the material deposited earlier was removed again.

Since streamflow amounts did not differ appreciably between the two periods, the recent removal of sediment from the river bed reflected decreased rates of sediment inputs to the system. Thus there appear to be phases of rapid aggradation associated with extreme events followed by longer periods of relatively slower removal (Goswami 1985).

It follows that one should be careful when interpreting changes in sediment yield over time at a particular gauging station and not immediately ascribe any increases to "deforestation" further up in the basin and vice versa (e.g. Hardjono 1980; Narayana 1987).

Obviously, any predictive equations of stream sediment load that fail to take into account extreme events related to mass wasting induced by tectonic or cyclonic activity are bound to produce more or less gross underestimates.The fact that a considerable fraction of material delivered to the streams by deep-seated mass wasting tends to be rather coarse and as such will be transported mainly as bedload (Carson 1985; Simon & Guzman-Rios 1990), aggravates the matter even further. Usually, the (generally unmeasured) fraction of the total load is assumed to be about 10 per cent (Delft Hydraulics 1989) although there are indications that this figure may be much higher in areas subject to intense mass wasting (Pickup et al. 1981; Simon & Guzman-Rios 1990).

As such, it will be no surprise to learn that actual rates of reservoir siltation in the Indian Himalayas (Gupta 1983), in the Philippines (Wooldridge 1986) and indeed in many other places (Mahmood 1987), severely exceeded predicted values, not so much because of "deforestation" as is frequently proposed (e.g. Murty 1985; Tejwani 1987, etc.) but rather as a result of inadequate data collection and processing (Amphlett 1988; Bruijnzeel & Bremmer 1989; cf. Hamilton & Pearce 1987).

With the above caveats in mind, what does research have to offer with respect to the influence of land management on basin sediment yields in the humid tropics?

As we have seen in the preceding sections, the presence or absence of a good vegetation cover or soil conservation measures is a strong determinant of amounts of sediment generated by surface erosion. Also, the occurrence of shallow mass movements may be increased considerably by the removal of tall vegetation. It follows that the overall effect of changes in land use on basin sediment yields will be strongly determined by the kind of processes supplying and removing sediment to / from the drainage network under natural conditions (Dunne 1984; Pearce 1986).

Clearly, in the case of high natural sediment yields as a result of steep terrain, high rainfall rates and geological factors, little, if any, influence will be exerted by man (Pain & Bowler 1973; Carson 1985; Whitehouse 1987). On the other hand, under more stable geological conditions and relatively low natural denudation rates, man-induced effects may be considerable (Van Dijk & Vogelzang 1948; Douglas 1967a; Bailly et al. 1974; Lam 1978; Dunne 1979; Fritsch & Sarrailh 1986). As a result, opportunities to reduce sedimentation rates will also differ greatly between cases as will be demonstrated by the following two examples.

Phewa Tal catchment (Central Nepal)

Available data for the 117 km^2 Phewa Tal catchment near Pokhara in the Middle Mountains of Nepal on which to base a sediment budget include estimates of basin sediment yield based on bathymetric surveys of Phewa lake (Impat 1981), rates of surface erosion associated with various types of land use (Impat 1981), and observations on mass movement processes (Ramsay 1985). Although the data has its limitations (see Bruijnzeel & Bremmer (1989) for details), an interesting picture arises from their combination.

On the basis of the bathymetric surveys Impat (1981) estimated the total amount of sediment trapped in the lake over the period 1976-1979 at about 33 t ha^{-1} yr^{-1} (26 m^3 ha^{-1}). Assuming a trap efficiency of 90 per cent (Dunne 1977), this would correspond with an average inflow of sediment into the lake of about 37 t ha^{-1} yr^{-1}, a figure that is quite comparable to other results obtained for this part of the Himalaya (Bruijnzeel & Bremmer 1989).

Adding up the respective contributions of surface erosion for the various land use types in the catchment, Impat (1981) also derived a basin-wide estimate of on-site erosion of 89,000 t yr^{-1}. Applying a sediment delivery ratio (SDR) of 0.3 (a reasonable value for a basin this size: Walling 1983), this would imply that of the 430,000 tonnes (37 t ha^{-1} times basin area) of sediment entering the lake each year, about 27,000 tonnes (0.3 times 89,000 t) or 6 per cent would be contributed by surface erosion. The remainder (about 400 000 tonnes) would have to be supplied by (riparian) mass movements and gully erosion.

Interestingly, on the basis of independent observations of landslide volumes and frequencies in the Phewa valley, Ramsay (1985) arrived at a total volume of sediment produced by various kinds of landsliding of ca. 310,000 m^3 yr^{-1}. According to Ramsay, about 90 per cent of this material was supplied by a few large failures in groundwater discharge zones exhibiting high transport efficiency and therefore responsible for a high proportion of overall sediment movement into the valley bottom river system (Ramsay 1987a; cf. Brunsden et al. 1981).

Although one can only guess as to what fraction of the material thus arriving at the valley floor is transported more or less directly to the lake, the similarity in volumes of sediment generated annually by these large slope failures and those deposited in the lake is striking indeed.

The finding that surface erosion and shallow mass movements contribute only a small fraction of the total stream sediment load in this particular environment has profound implications for the downstream benefits that can be expected from a catchment rehabilitation programme (Hamilton 1987).

Fleming (1988) carried out a tentative computation to determine the effect on the siltation rate of the lake of reducing soil losses from overgrazed land in the basin to a level associated with improved pastures (Impat 1981). Not surprisingly in the light of the above considerations, the effect was a negligible reduction in lake siltation of ca. 1 per cent (Fleming 1988). Likewise, when Carson et al. (1986) carried out a similar analysis for the Kali Gandaki river basin in West-Central Nepal (11,138 km^2), they arrived at a reduction of 7 per cent in basin sediment yield following rehabilitation measures.

This of course does not mean that restorative measures should not be taken. They definitely should, but rather in view of reducing losses of productivity on cultivated fields and degraded grassland (Shrestha 1988).

Konto catchment, East Java, Indonesia

A reasonably comprehensive study of the sediment dynamics of the 233 km^2 Konto basin in the volcanic uplands of East Java was recently carried out by Rijsdijk & Bruijnzeel (1990). Available data include two years of suspended and bed sediment yields at six gauging sites representing forested headwater catchments and intensively used hills and plains within three major landscape units in a nested set-up (cf. Amphlett 1988), as well as measurements of surface erosion in degraded forest and scrubland, coffee plantations, terraced dry land agricultural land, settlements and various types of roads and trails. In addition, several active gully systems were monitored for runoff and sediment and landslide occurrence was mapped at the end of two consecutive rainy seasons along a 36-km transect. Finally, limited observations were made of bank retreat rates (Rijsdijk & Bruijnzeel 1990). An artificial lake situated at the outlet of the Konto basin exhibited an average siltation rate of about 232,000 m3 (1977-1988; PU Brantas 1989) or about 10 m^3 ha^{-1} (cf. the 26 m^3 ha^{-1} quoted above).

The three forested headwater basins (340-1170 ha) exhibited considerably different sediment yields, both in terms of suspended sediment- and bed-loads, which mainly reflected differences in geological substrate. Low sediment yields (0.23-1.15 t ha^{-1} yr^{-1}) were recorded for basins underlain by bouldery and lava deposits whereas higher values (ca. 3.8 t ha^{-1} yr^{-1}) were observed for streams draining unconsolidated "lahar" deposits whose banks easily collapsed.

By contrast, the densily populated areas immediately downstream typically produced 21-26 t ha^{-1} yr^{-1}, mostly generated by surface erosion from built-up areas, roads and non-irrigated fields and to a much lesser extent by bank erosion. Somewhat surprisingly, hillside mass wasting contributed little sediment (ca. 1 per cent) to the overall total (Rijsdijk & Bruijnzeel 1990). Elsewhere in Java, much higher sediment yields have been reported for forested uplands underlain by marly deposits prone to mass wasting (Van Dijk & Ehrencron 1949; Delft Hydraulics 1989).

Interestingly, ratio's of total stream sediment loads to overall sediment inputs by surface and bank erosion (i.e. SDR) in the Konto area were quite high (ca. 0.5 to 1.1), at least compared to values predicted for basins of this size by a frequently used relationship derived by the US Soil Conservation Service (Walling 1983). This suggested relatively efficient removal of material by the streams of this upland volcanic area.

Values of SDR were quite different for the two years of observation (see Rijsdijk & Bruijnzeel (1990) for details), illustrating the difficulty of capturing such a complex phenomenon as catchment sediment delivery in a single variable (Walling 1983), and pointing to the importance of long-term observations of the processes involved (Dunne 1984). Nevertheless, the major sources of sediment in the Konto area could be identified and to a fair extent be quantified. The data suggest that considerable reductions in basin sediment yields could be obtained by improving the drainage from settlements and other impervious surfaces and by the construction of sediment traps at strategic locations in such areas.

As such, there appears to be a wide variation in the magnitude of downstream effects of on-site rehabilitation measures, depending on the overall geological and climatic setting of an area. Again, in regions with relatively low natural erosion rates there is considerable scope for measures aimed at reducing impacts of logging, clearing and conversions.

A number of countries with tropical forests have produced detailed guidelines for minimising erosion and sedimentation effects of timber harvesting (e.g. Cameron & Henderson 1979; Abdul Rahim 1984).

These include comprehensive planning of locations for such notorious sediment producing surfaces as skid tracks, landings and haulage roads (cf. Megahan 1977; Rothwell 1978; Adams & Andrus 1990), of equipment to be used, the timing of an operation with respect to rainfall and hence soil moisture status (Martin 1970; Van der Weert 1974), and the conservation of riparian buffer strips (Bosch & Hewlett 1980; Clinnick 1985).

The value of such measures has been demonstrated by Gilmour (1971) and Baharudin (1988) in Queensland and Malaysia respectively. However, for a buffer strip to be effective, it needs to include all tributary gullies, as demonstrated by the work of O'Loughlin et al. (1980). Even so, minimising the flow of water from road surfaces, etc. into such gullies remains important.

Naturally, all of the above measures also pertain to clearing operations, especially in the case of urbanisation (Chinnamani & Sakthivadivel 1985; cf. Lootens & Lumbu 1986). In addition, depending on the type of conversion, specific conservation practices will be needed (see Pearce & Hamilton (1986) for an excellent summary). For example, in the case of extractive tree plantations, harvest trails need to be laid out carefully (Maene et al. 1979) whereas in the case of pastures, fire and grazing intensities will need specific attention (Jasmin 1975; Dano 1990).

Finally, there is an extensive literature on soil conservation practices for dry land agriculture (Hudson 1971; Lal 1983; Morgan 1986).

Whatever the kind of measures taken, it should be realised that effects will be felt most readily on-site and that these tend to become smaller for larger basins as a result of the inverse relationship between basin size and sediment delivery ratio (Walling 1983; Fritsch et al. 1987; Rijsdijk & Bruijnzeel 1990).

A related and often overlooked aspect is the time scale at which any downstream benefits from upland watershed management activities are likely to become noticeble (Pearce 1986). This is clearly illustrated by the fact that the sudden inputs of sediment mobilised during the 1950 earthquake in Assam remained detectable in the sediment load of the Brahmaputtra for more than twenty years (Goswami 1985).

As pointed out by Pearce (1986) and Hamilton (1987), there could be very little change for decades in the amounts of sediment carried by major rivers in their lower reaches, even if all man-induced erosion in the uplands could be eliminated at once. The reason for this lies in the fact that there is so much sediment (both from previous man-caused and natural erosion) stored in the system, that this effectively forms a long-term supply (cf. Dietrich & Dunne 1978; Trimble 1981).

This contention was also supported by the results obtained for a major land and stream rehabilitation programme in China, which indicated that under prevailing environmental conditions, reductions in sediment yield of up to 30 per cent could be expected for catchments up to 100,000 km^2 after about two decades only (Mou 1986).

Clearly, the frequently voiced claim that upland reforestation etc. will solve most downstream problems does require some specification of the spatial and time scales involved. As pointed out by Hamilton & Pearce (1987), it is important not to raise unrealistic expectations in this respect. Otherwise the credibility of watershed management and environmental planning professionals and possibly years of progress towards more rational land use could be lost.

One of the most serious consequences of erosion is the decline in agricultural productivity which in some upland areas is already severely threatening the livelyhood of upland farmers and their families (Carson 1985; Shrestha 1988). Indeed, a number of conversions of forest to other land use types in the tropics does not seem to be sustainable without large inputs of fertilisers, etc.

The next chapter will examine the aspect of soil productivity in somewhat more detail, distinguishing between the three levels of intensity of disturbance recognised in section 4.1, viz. low, medium and high.

5 HYDROCHEMICAL AND SOIL CHEMICAL RESPONSES TO (DE)FORESTATION

5.1 Introduction

As suggested by Herrera et al. (1978a), forests on highly depleted soils in the humid tropics are only able to maintain themselves at relatively high biomass levels (typically up to about 450 t ha^{-1}; Klinge 1976; Ohler 1980) by various nutrient conserving mechanisms.

Probably the most important of these is a thick root mat at the surface which has been shown to be an extremely efficient filter for nutrients reaching the forest floor in crown drip, stemflow, and litterfall (Figure 10b; Stark & Jordan 1978; Herrera et al. 1978b). A similar root mat, albeit less conspicuous than that of Amazonian forests, may be observed in forests on substrates in which a certain key nutrient (e.g. potassium on ultrabasic rocks) is extremely scarce (L.A. Bruijnzeel & M.J. Waterloo, unpublished data).

As demonstrated in chapter 3, the various conservation mechanisms tend to produce a relatively "tight" nutrient cycle with only small amounts of nutrients being lost from the system in the drainage water. Forests on more fertile substrates, on the other hand, will show a more "open" type of nutrient cycle (Baillie 1989; Bruijnzeel 1989b). As shown by the data presented in Table 3, various levels of soil fertility each exhibited a characteristic nutrient export pattern.

Upon clearing tropical forests substantial increases in total water yield have been observed (Table 4). As such, a (much) larger volume of water may (temporarily) percolate through the soil after forest removal (Figure 18). Naturally, oligotrophic sites will be more vulnerable to disturbance than eutrophic sites since this often brings about the partial or complete disruption of the nutrient conserving root mat (Jordan 1985). In such cases, extra losses of precious nutrients through increased leaching, apart from those removed in harvested biomass and burned residues, are to be expected, leaving the system as a whole even more impoverished (Brinkmann & Nascimento 1973; Russell 1983). In addition, there is the danger of increased nutrient losses via surface erosion, especially after the use of heavy machinery (Kang & Lal 1981; Bruijnzeel & Wiersum 1985; cf. sections 4.1 and 4.5.1).

In the following we will examine the hydro- and soil chemical effects of forest disturbance and conversion, making use of the fact that amounts of nutrients carried in headwater streams or drainage water constitute a good indicator of an area's nutrient status (cf. section 3.1).

5.2 Response to low-intensity disturbances

Old-growth tropical forests are highly dynamic ecosystems in which tree mortality is roughly balanced by growth and approximately five per cent of the forest may be in gap phase at any one time (see Uhl et al. (1988a) and several references therein). Wind gusts at the onset of intense rain storms are probably one of the most important natural disturbance agents in equatorial terrain of low relief (Uhl 1982) and may occasionally knock down whole sections of forest, thereby uprooting many of the large trees and damaging smaller ones (Uhl et al. 1988a).

Generally, several micro-habitats are distinguished in gaps (Brokaw 1982), viz. the "trunk zone" (the area below and immediately next to the downed bole), the "crown zone" (the debris-laden area where the crown fell, usually outside the gap opening), and the "open zone" (the area between the fallen trunk and the edge of the gap). Theoretically, one might expect topsoil nutrient concentrations in the trunk and crown zones to increase steadily following the sudden addition and subsequent decomposition of large amounts of organic matter to the forest floor. Also, temporally decreased evapotranspiration in the open zone (Luvall 1984) might induce increased percolation and hence nutrient leakage.

However, results of recent experimental studies indicate otherwise. For example, Vitousek & Denslow (1986), working on a moderately fertile volcanic soil in Costa Rica, were unable to detect statistically significant differences between available nitrogen and phosphorus levels of the crown zones of downed trees and closed-canopy forest. Similarly, Uhl et al. (1988a), studying forest dynamics on highly depleted Oxisols in Venezuela, did not find any difference in soil nutrient concentrations between the trunk and crown micro-habitats and the open zone or the intact forest. Also, there was no apparent relationship between rate of bole decomposition and soil nutrient levels, although there were statistically significant differences in soil fertility between gaps (Uhl et al. 1988a).

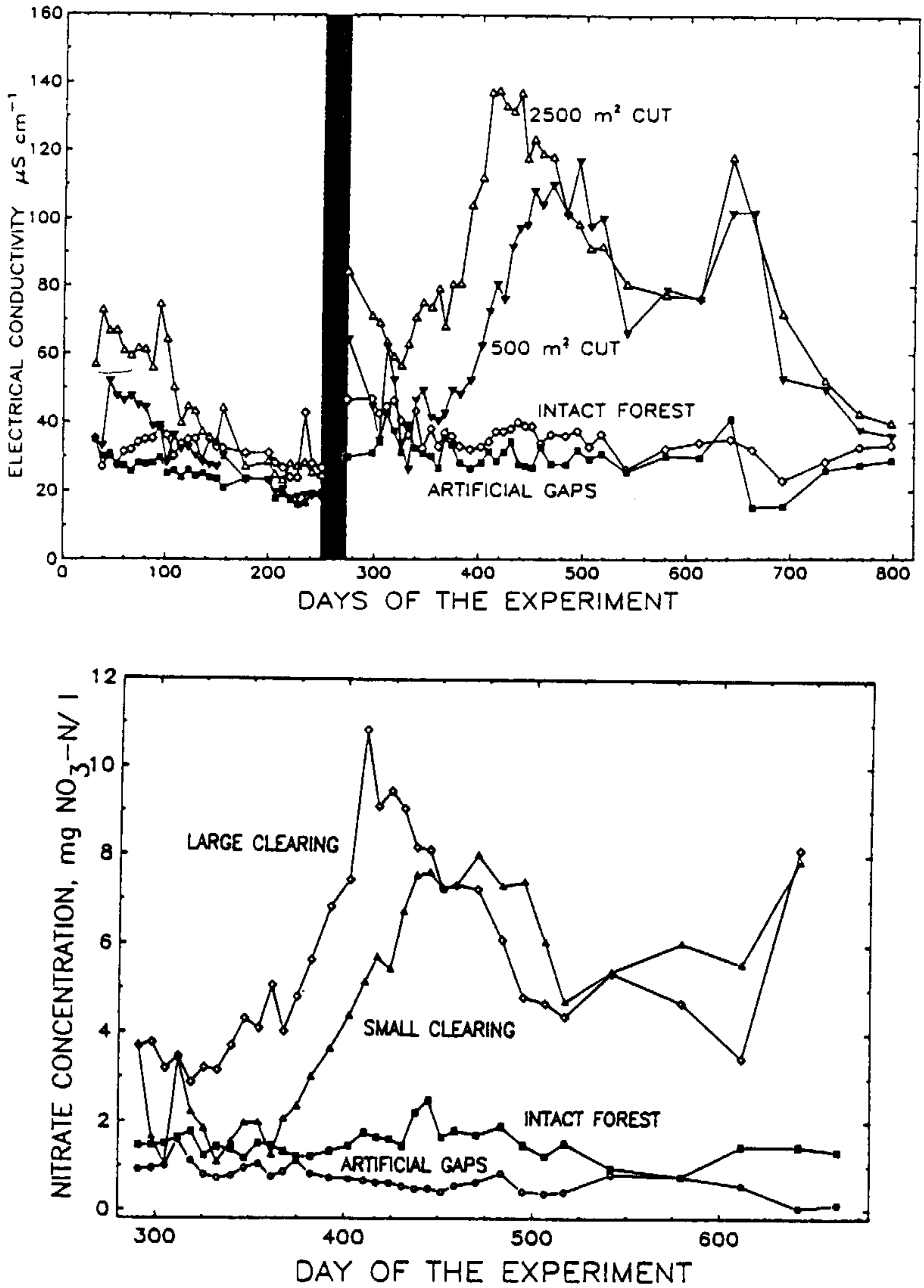

Figure 27. Temporal trends of (a) electrical conductivity and (b) volume-weighted nitrate concentrations in soil water under undisturbed forest, artificial gaps and clearings in Costa Rica (after Parker 1985).

One could argue that the monitoring of soil water quality rather than soil chemical changes would provide a more direct and sensitive estimate of leaching losses than the use of standard soil analytical techniques (Verstraten 1980; Bruijnzeel 1983a). Yet, neither Parker (1985) nor Uhl et al. (1988a) recorded significant increases in nutrient concentrations of leachate water following the creation of small gaps (47-164 m^2) on either of the above soil types (Figure 27).

It must be concluded that nutrients released from decomposing treefall debris (mainly leaves) were taken up quickly enough by the surrounding vegetation to prevent extra leakage. Naturally, as long as the root mat (if one is present) remains intact (which is usually the case in natural gaps: Sanford (1985) in Uhl et al. 1988a) and gaps are relatively small, this result was to be expected.

However, natural gaps may be much larger than these experimentally created ones (Uhl et al. 1988a). What then would be the critical size of a gap beyond which significant nutrient leakage would occur? Apart from the single-tree gaps already referred to, Parker (1985) also studied percolation rates in plots of 500 m^2 (10 x 50 m) and 2500 m^2 (50 x 50 m) before and after clearance (Figures 18 and 27). The results clearly showed large increases in leachate concentrations (determined at -70 cm with suction lysimetry) after about four months following clearcutting in both cases. The pulse of elevated concentrations reached a maximum after six to eight months and lasted for about a year. Interestingly concentrations started to rise earlier in the 2500 m^2 plot which also experienced higher soil moisture levels and therefore higher (unsaturated) hydraulic conductivities (Figures 18 and 27; Parker 1985).

Ions which showed the strongest elevations in concentration after cutting were nitrate (an average three- to fourfold increase) and hydrogen (1.7 to 2.5 times), whilst concentrations of calcium and magnesium almost doubled. Potassium did not respond markably, nor did phosphate or ammonium (Parker 1985). The investigator suggested that the low ammonium levels present in the soil water reflected a mixture of rapid nitrification and uptake by the regenerating vegetation (cf. Robertson (1984) who reported ammonium availability to be the main regulator of nitrification rates in young secondary forests at the same location). No rise in nutrient concentrations of streamwater was recorded during the 404-day post-cut period (Parker 1985).

5.3 Response to disturbances of intermediate intensity

5.3.1 Selective logging

It was shown in the preceding section that the release of nutrients from decomposing fallen trees was more or less matched by the uptake by surrounding vegetation, thus avoiding serious leakage of nutrients from the ecosystem. However, one may wonder whether the sudden addition of large amounts of organic debris to the forest floor during logging operations (De Graaf 1986; Phillips 1987) and the associated release of nutrients via decomposition will not exceed the nutrient uptake/retention capacity of the remaining forest, especially so on sandy soils (Poels 1987).

Studies of the hydrological effects of selective logging in rain forest areas have been conducted in Queensland (Gilmour 1977), Surinam (Poels 1987) and Peninsular Malaysia (Abdul Rahim et al. 1985; Abdul Rahim 1989; Zulkifli Yusop 1989). The latter two studies also reported on the associated nutrient exports, whilst Gillman et al. (1985) investigated changes in soil nutrient content during the first four years after logging rain forest in Queensland.

Gilmour (1977) did not observe any statistically significant changes in streamflow patterns following logging, adding that this was probably due to the rather extensive character of the type of logging practised in the area, which leaves a fair amount of canopy (and presumably also the forest floor) in tact (Cassells et al. 1984; Gillman et al. 1985).

Studying the effects of selective logging on soil chemical properties in a somewhat drier area underlain by granite and to the north of Gilmour his study catchment, Gillman et al. (1985) reported statistically significant decreases for all parameters in areas of snig track depressions (with top-soil removed by bulldozing, affecting 13 per cent of the area), increases in the mounds formed at the sides of the snig tracks (15 percent of the area), and no significant differences in areas with decomposing slash (but little or no soil disturbance; 40 per cent of the area) or in areas of increased irradiation following opening of the canopy (16 per cent of the area). Gillman et al. (1985) concluded on the basis of a very well-planned stratified sampling programme that amounts of nitrogen and exchangeable cations over the whole area (5.6 ha) had not altered.

However, there was a loss of organic carbon of about 15 per cent over the four-year study period (Gillman et al. 1985).

Conversely, extracting ca. 40 per cent of the stocking from a hill dipterocarp forest on granitic terrain at Bukit Berembun in Peninsular Malaysia by means of the "san tai wong" (winch lorries) method of logging produced significant increases in streamflow (Table 4; Abdul Rahim 1989). Corresponding increases in water yield for an adjacent small catchment subjected to "supervised" logging were considerably smaller (Abdul Rahim 1990; see footnotes of Table 4 for details; cf. Thang 1986).

The losses of nutrients associated with the two treatments were described by Zulkifli Yusop (1989). He reported a 300 per cent increase in nitrate concentrations during the first year after the cut in the commercially logged area (C1), and a more moderate 180 per cent increase for the catchment with supervised logging (C3). Nitrate concentrations dropped back to their pre-logging values after the first year (cf. Parker 1985). The corresponding increases in exports of nitrate amounted to 465 (C1) and 155 (C3) per cent of the values predicted from the forested control basin, although the absolute amounts involved were rather unimpressive at ca. 2 and ca. 0.65 kg ha^{-1} yr^{-1} in C1 and C3 respectively.

Similarly, potassium concentrations in streamwater emerging from C1 increased by some 70 per cent during the first year and remained ca. 30 per cent higher than predicted during the second year, whilst no statistically significant differences were observed for C3 in this respect. Corresponding increases in exports amounted to 167 (C1) and 47 per cent (C3) with the latter value mainly reflecting the increase in streamflow.

Conversely, concentrations of calcium and magnesium did not respond very much to either treatment (Zulkifli Yusop 1989), although exports of both elements increased temporarily due to the increase in streamflow following the intervention. Monthly nutrient loads exported from C3 were decidedly smaller and returned more rapidly to pre-logging levels than those associated with the commercial logging treatment. In the latter case, extra amounts leached from the system during the first two years after logging corresponded roughly with equivalent nutrient inputs via bulk precipitation (as measured at nearby Pasoh: Manokaran 1980), of one (calcium), two and a half (potassium) and four years (magnesium) (Zulkifli Yusop 1989).

Although the Bukit Berembun study is the most rigorous study of hydrochemical response to logging of tropical forest published to date, the paired catchment technique remains essentially a "black box" approach which needs to be supplemented with process studies if we are to understand more fully what is happening in the various compartments of the ecosystem (Bruijnzeel 1989b). It is encouraging, therefore, to note that more recent experiments in Malaysia (cf. Malmer 1990) and Indonesia (cf. Bons 1990) have adopted this combined approach.

An interesting experiment on the effects of "refining" a selectively logged forest in Surinam by poisoning non-commercial species and cutting lianas in order to promote the growth of merchantable species (De Graaf 1986) has been presented by Poels (1987). He sampled two forest streams (on a weekly basis) for four years, one draining undisturbed rain forest ("East Creek", 140 ha), the other draining an area which had been logged selectively one to two years before the start of the sampling programme ("West Creek", 155 ha). The logging operation was estimated to have killed or removed ca. ten per cent of the vegetation. After almost two years of sampling, the "West Creek" forest was subjected to refinement, whereby an estimated further 40 per cent of the trees were killed. Streamflow from the total area (295 ha) was determined at a weir below the junction of both creeks throughout the sampling period and outflows of water from the two areas were assumed to be equal, at least for the sake of computations (Poels 1987). No calibration of nutrient concentrations between basins prior to the refinement was performed.

It will be clear that a proper analysis of data produced by the above set-up will be difficult (Hewlett & Fortson 1983; Ibrahim & Chang 1989), also because of the considerable variation in rainfall that occurred during the study period (Poels 1987). Nevertheless, a comparison of average concentrations in streamflow for the two areas before the refinement of "West Creek" revealed that calcium levels in the latter were higher by about 15 per cent, those for magnesium by seven per cent, and for potassium by ca. 29 per cent. These differences may reflect the release of nutrients from decomposing slash (Ewel et al. 1981; Zulkifli Yusop 1989), although the nature of the experiment prevents an examination of other factors such as slight differences in weathering rates of the underlying granitic rocks (cf. MacKay & Robinson 1987).

The effects of the refinement operation, which need to be superimposed on the already existing (and probably gradually changing) differences in streamwater quality, were a short-lived increase (first year only) of potassium concentrations (ca. 30 per cent), a twenty per cent rise in calcium concentrations and a modest five per cent increase for magnesium levels (with the latter two values pertaining to the two years after refinement). Naturally, these estimates are approximations and it is likely that actual losses brought about by the refinement will be larger than supposed by Poels (1987) in view of the already mentioned assumption of unaltered flow. As shown by the Malaysian example, a fair rise in streamflow following partial removal of the vegetation can be expected (Abdul Rahim 1989).

Nevertheless, absolute amounts of nutrients leached from undisturbed and treated forests were small by most standards (cf. Table 3), usually in the order of a few kg ha^{-1} (Poels 1987), and it must be possible to have these compensated for by atmospheric inputs of nutrients within a few years (Poels 1987; cf. Bruijnzeel 1989a).

Chemical analysis of soil samples taken shortly before and two years after refinement suggested rather large losses of adsorbed macro-nutrients down to a depth of 120 cm (Poels 1987). These losses could only partially be explained in terms of uptake by the regenerating vegetation (equal to about 40 per cent of amounts released from logging debris), and certainly not by the quantities reportedly lost via streamflow. Further work of a more stringent nature on this important subject is desirable (cf. the sampling programme of Gillman et al. (1985) which was stratified according to five levels of disturbance).

Of course, increased nutrient losses following logging also occur in the form of the often observed enhanced sediment loads of forest streams (Douglas 1967a; Gilmour 1977; Fritsch 1983; DID 1986; Baharuddin 1988). However, chemical analyses of particulate material (both organic and inorganic) carried by streams in tropical forest areas are rare (Brinkmann 1985; Lewis 1986). Also, in the absence of process studies it is difficult to establish relative contributions from various sources, be it runoff from logging tracks or landings (Ruslan & Manan 1980), erosion of disturbed riparian zones and ephemeral channels (Bons 1990; Greer et al. 1989), or enhanced bank erosion associated with increased peakflows after clearing (Kaatee 1989). More work is needed in this respect.

5.3.2 Forest fires

Until recently, forest fires were not considered a very important phenomenon in tropical rain forest areas. However, the great fires occurring in Borneo in 1982 and 1983, which damaged more than four million ha of forest (Malingreau et al. 1985), and the deliberate burning of millions of hectares of forest along the southern fringe of Amazonia in 1987 and 1988 for the creation of pastures (Whitmore 1990) have forced a revision of this view.

The Borneo fires were believed to be the result of a combination of natural and human factors. A severe drought preceding the fire led to the shedding of leaves and to the accumulation of dry litter on the forest floor (cf. Proctor et al. 1989). In addition, selectively logged areas provided both extra fuel in the form of dried out logging debris and improved access via logging trails (Phillips 1987). As such, logged-over forests suffered more than primary rainforest (Malingreau et al. 1985).

The effects of fire on the regulation of amounts and quality of streamflow in forested areas are well documented (e.g. Anderson et al. 1976), although studies pertaining to the humid tropics are few in number. Of course, there is the inherent difficulty in the study of hydrological impacts of forest fires that one either does not have information on pre-fire hydrological behaviour or that one's gauging stations are destroyed by the very fire. Usually, therefore, flow data are not available for the first year or so after the event (Brown 1972; Kusaka et al. 1983), during which period losses of nutrients can be expected to be the most severe (Uhl & Jordan 1984; MacKay & Robinson 1987). Arguably, the best way of overcoming this difficulty is through experimental studies (O'Loughlin et al. 1982; Uhl et al. 1982; Uhl & Jordan 1984).

Losses via erosion and increased leaching depend strongly on post-fire climatic conditions. Leitch et al. (1983) described a case in south-eastern Australia where a single rainstorm of moderate magnitude, occurring six days after a wildfire had raged through a eucalypt forest, produced widespread sheet erosion and considerable scouring of gullies (cf. Brown 1972). Interestingly, the fresh ash layer was found to be highly absorptive, but the underlying soil had become hydrophobic down to a depth of 10 cm, both as a result of the fire and a severe drought preceding it.

The excessive hydrological response of the burnt area to only a moderate amount of rainfall was interpreted in terms of this strongly reduced infiltration capacity of the soil. It took more than three months before the topsoil had resumed its normal absorptive properties, presumably via capillary wetting from below (Leitch et al. 1983). In addition, it was estimated that at least 22 t ha^{-1} of ash and burnt topsoil, containing ca. 13 and 18 per cent of the pre-fire nitrogen and phosphorus contents of the biomass respectively, were removed from the catchment during this particular event (Leitch et al. 1983).

Naturally, losses via surface erosion may be much greater under conditions of intense tropical rainfall and steep terrain (Toky & Ramakrishnan 1981; Hurni 1982; Sato et al. 1984; cf. the section on shifting cultivation).

To the best of my knowledge, nutrient losses associated with increased leaching following natural forest fires in the tropics have not been published. However, Grip (1986) reported electrical conductivity values of streamflow emerging from forests in Sabah that had burnt down two years before to be still more than twice those for adjacent streams draining (selectively logged) forest not affected by the great fires of 1983.

Similarly, Nakane et al. (1983) found losses of potassium and magnesium from small catchments in the subtropical belt of Japan, whose forests had been burnt a year and a half before the start of the observations, to be significantly greater than the corresponding losses from the forested control catchment. The increased exports were partly due to the increase in water yield (32 per cent) following the fire (Nakane et al. 1983).

Although the above examples show that effects of forest fires on water quality in the (sub)tropics may remain detectable for quite some time after the event, they do not give information on the concentrations of nutrients in streamflow during the critical period between the fire and the re-establishment of vegetation.

The only studies providing this type of information seem to be the experiments of Uhl et al. (1982) and Uhl & Jordan (1984) who monitored the nutrient dynamics of 0.5 ha plots after cutting and burning so-called "Caatinga" and "Tierra firme" rain forests on very infertile Spodosols and Oxisols respectively in southern Venezuela.

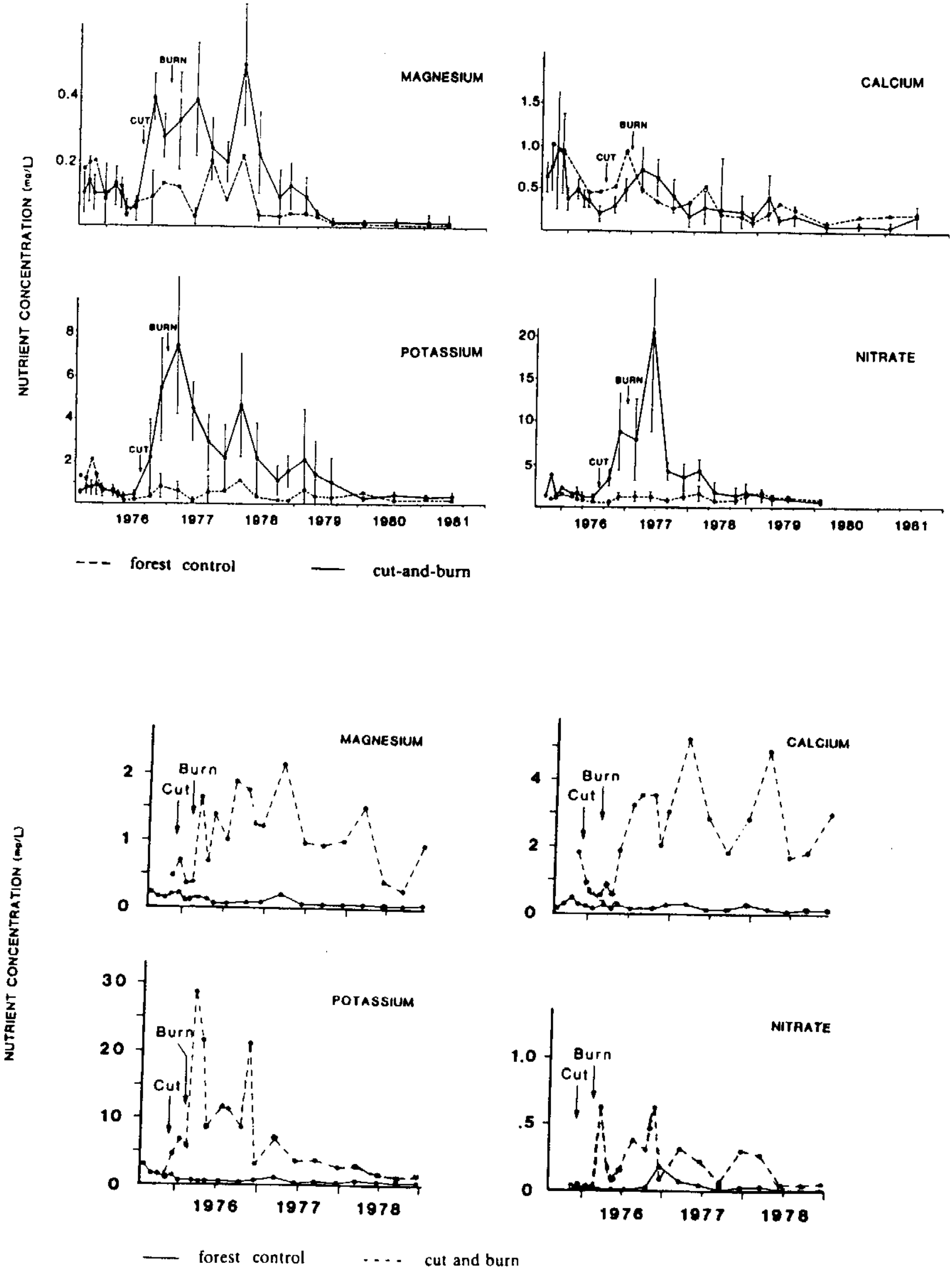

Figure 28. Elemental concentrations in free-draining soil water at 40 cm depth in undisturbed and adjacent cut-and-burned forest plots at San Carlos de Rio Negro, Venezuela: (a) Oxisol site (after Uhl & Jordan 1984) and (b) Spodosol site (adapted from Uhl et al. 1982).

The chemical composition of drainage water as collected by means of zero-tension lysimeters (cf. Figure 11a) was monitored for a number of years, both in the disturbed sites and in undisturbed control plots (Figure 28). At the Oxisol site (Figure 28a) concentrations of magnesium, potassium and nitrate in the leachates rose quickly after the burn, but subsided to pre-burn levels after about two years. Interestingly, calcium reacted less strongly (Figure 28a). Unfortunately, **amounts** of nutrients leached from the ecosystem were not given explicitly by Uhl & Jordan (1984).

The response to disturbance at the nearby Spodosol site was somewhat different (Figure 28b). Calcium responded much more vigorously now and nitrogen constituents much less than at the Oxisol site. Losses of potassium were again much greater than for any other element, but the pulse was rather more short-lived, which is in line with the sandier texture of the Spodosol. The fact that "Caatinga" forest is thought to be limited in nitrogen (Vitousek & Sanford 1986) might account for the moderate increase in nitrogen concentrations in drainage water after burning (cf. Sollins & McCorison 1981). Again, total amounts of nutrients leached from the system during the operation were not presented.

Nevertheless, the reported changes in concentrations at both sites are most valuable and could perhaps be used at a later stage when our understanding of water use by regenerating vegetation will have improved sufficiently to model changes in drainage rates after tropical forest clearing (cf. Table 4).

5.3.3 Shifting cultivation

Although the practice of shifting cultivation (see section 4.1) is essentially based on the same processes all over the tropics, the outcome of the cycle may differ between locations, depending on the interaction of climate and soil, manifesting itself both in site productivity (regenerative potential) and in the leaching regime of a soil (Andriesse 1987).

The decline in yields during the cropping phase is believed to be a result of soil fertility depletion, increased weed infestation, deterioration of soil physical properties (e.g. erosion), increased insect and disease attacks, and social factors (Sanchez 1976). In the following we will concentrate on the soil fertility aspect.

The minimum duration of the fallow period needed to restore topsoil fertility obviously depends on climatic and edaphic factors, with longer periods of rest needed in areas of high erosion and leaching potential. For example, the Lua tribe in northern Thailand (annual rainfall 1500 mm) has developed a cycle of one crop of upland rice followed by a forest fallow of nine years (Zinke et al. 1970). Hatch (1983) on the other hand, observed crop yields in perhumid Sarawak to be declining after shortening the regenerative phase from ca. 15 years to seven or even less. Scott (1987) described a case in Peru where repeated burning of secondary vegetation had led to the formation of grasslands with very low nutrient contents in both vegetation and soil.

Soil nutrient levels are subject to a number of changes during clearing, burning, cropping and fallowing (Figure 29a). During the drying phase, some nutrients are added to the topsoil by decomposition of the slash (Ewel et al. 1981; cf. preceding sections). Much larger amounts of nutrients however are released upon burning the dried material (Nye & Greenland 1960, 1964; Seubert et al. 1977; Ewel et al. 1981; Stromgaard 1984). Naturally, the quantities depend on the biomass of the forest (age, fertility of substratum; Bartholomew et al. 1953; Fölster et al. 1976; Jaffré 1985) as well as the intensity of the burn (Hatch 1983; Andriesse 1987).

Contrary to common belief, not all of the nitrogen is lost upon burning (Seubert et al. 1977; Ewel et al. 1981; Stromgaard 1984), although there is a strong relationship between volatilisation losses and maximum temperatures during burning (Andriesse 1987). In addition, a considerable proportion of the fresh ash may be blown away in some areas by wind gusts before the onset of the rains (Toky & Ramakrishnan 1981).

Since burning in the wet tropics is often incomplete (Laudelout 1954; Seubert et al. 1977; Andriesse 1987; DID 1986), nitrogen and other nutrients may continue to be supplied from decomposing stems, branches and roots for quite some time (Jordan 1987; Uhl 1987; Buschbacher et al. 1988).

The release of bases from the ashes raises the pH of the soil (Figure 29a), the extent being governed by initial soil acidity and amount of ash. The duration of the effect differs strongly between sites as a function of rainfall regime and edaphic characteristics (Sanchez 1976). For example, Nye & Greenland (1964) found an initial increase from 5.2 to 8.2 in the top layer of an Alfisol in Ghana (moderate rainfall) after

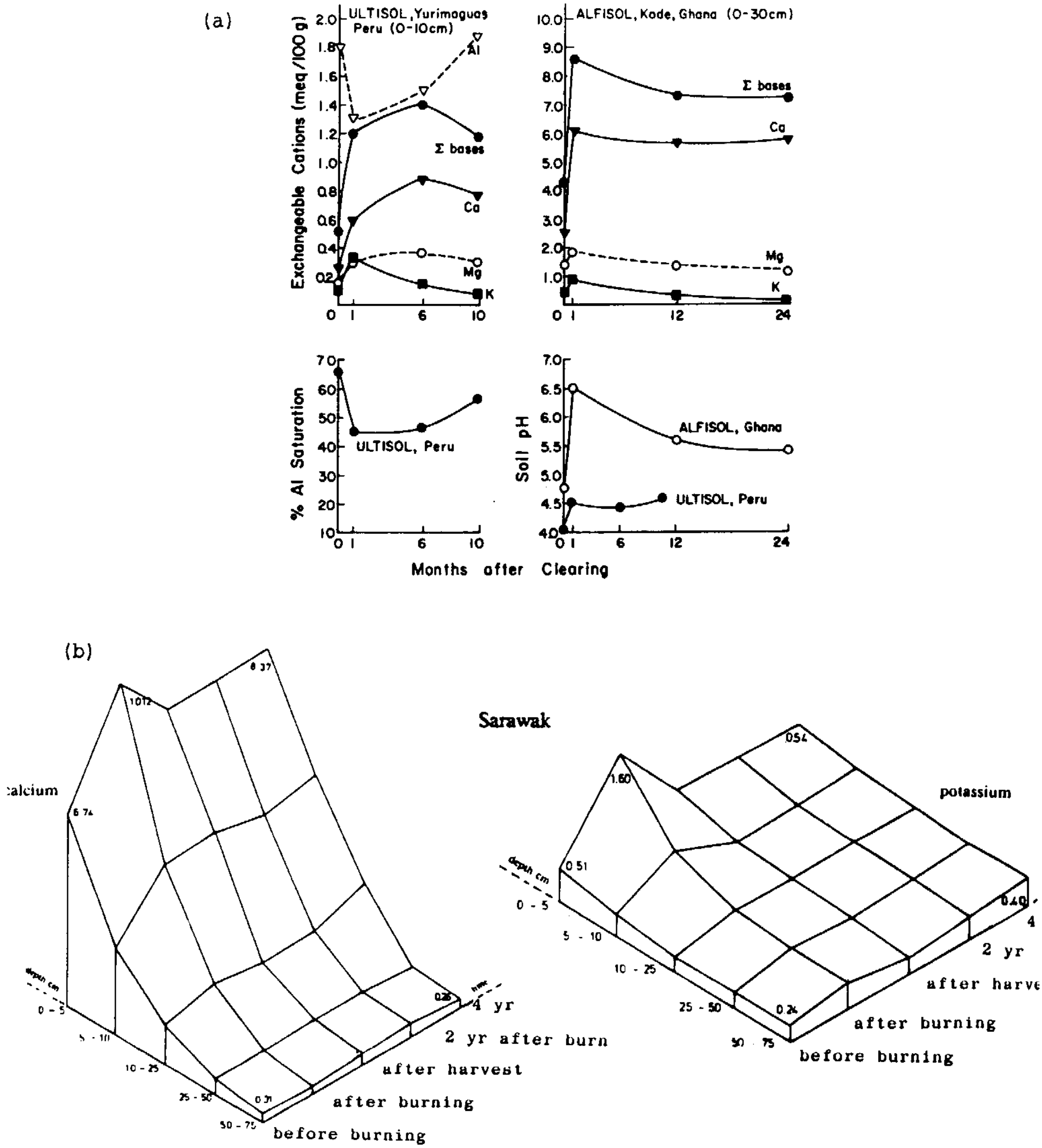

Figure 29. (a) Changes in exchangeable bases and soil acidity following clearing and burning the forest at an Ultisol site in Peru and at an Alfisol site in Ghana (after Sanchez 1976);
(b) Changes in exchangeable calcium and potassium (meq.100 g^{-1}) at various depths in the course of shifting cultivation on a poorly drained Ultisol in Malaysia (after Andriesse 1987).

burning a 50-year-old secondary forest. Some two years later, topsoil pH still amounted to 7. Conversely, only a modest increase in pH (from 3,8 to 4.5) was observed after burning the forest on highly depleted Oxisols under a more aggressive rainfall regime in central Amazonia, whilst the increase disappeared within four months (Brinkmann & Nascimento 1973).

Similarly, changes in calcium, magnesium and potassium concentrations during the cropping (and regrowth) phases may vary considerably between sites, again reflecting differences in initial soil fertility, nutrient uptake, erosion and leaching regimes (Zinke et al. 1970; Aweto 1981; Toky & Ramakrishnan 1981; Figure 29a,b). However, there is a general trend for potassium (and sodium) to disappear much more rapidly than either calcium or magnesium (Figure 29a), suggesting the dominance of leaching in removing nutrients from the (top)soil.

Nevertheless, as shown in Figure 29b, some of the nutrients washed out of the topsoil may be retained in the subsoil, e.g. through fixation by clay minerals (Andriesse 1987), and may therefore not be lost to the regenerating forest. A similar point relating to nitrate-nitrogen was made by Matson et al. (1987) for a cleared site on volcanic soil in Costa Rica (see also Robertson (1989) for a recent review of nitrogen dynamics following tropical forest disturbance).

Phosphorus presents a special case in that considerable amounts may become immobilised after its release by the fire through complexation with aluminium and iron compounds in the soil (Sanchez 1976) rather than being leached from the profile (cf. Parker 1985). Andriesse (1987) has estimated that up to 30 per cent of initially present "active" phosphorus became fixed in this way during the cropping and early regrowth stages on his somewhat poorly drained Ultisol in Sarawak.

As such, even though reserves of total and organic phosphorus at the end of the cropping phase may exceed those before burning, the question remains to what extent these are available for plant growth. According to Sanchez (1976), the decline in available phosphorus during the cropping period may be one of the most important reasons for abandoning a field to forest regrowth. Phosphorus fixation is usually less pronounced in sandier soils (Sanchez et al. 1983).

Jordan (1987) and Uhl (1987) drew attention to the dramatic contrast in the ability of crops and natural successional vegetation to take up nutrients in situations of low soil fertility. According to them, features exhibited by successional species which permitted survival under the conditions of restricted nutrient availability typical of abandoned rain-forest fields included: (1) high root : shoot ratio's (i.e. high proportions of energy are allocated to developing root biomass), (2) high incidence of mycorrhizal infection (St. John & Uhl 1983), (3) ability to take up nutrients in extremely low concentrations (Haines et al. (1986) in Uhl 1987), and (4) efficient nutrient use (i.e. high carbon : nutrient ratio's; Vitousek 1984).

The high efficiency of nutrient uptake by successional vegetation at Jordan and Uhl their study site was illustrated by the fact that five years after the burn the forest regrowth contained 23 per cent of the phosphorus present in the pre-burn forest live biomass, 39 per cent of the potassium, 45 per cent of the magnesium, and 48 per cent of the calcium (Uhl & Jordan 1984; cf. Jaffré 1985). Also, by that time soil nutrient levels were not discernible anymore from those in the control forest, presumably as a result of the combined action of uptake and leaching (Uhl & Jordan 1984).

However, vegetation recovery, and hence nutrient accumulation, was much less after abandoning fields in the same area after six years of intense farming or after abusive clearing and farming methods (Uhl 1987; cf. sections 4.1 and 5.4).

5.4 Response to high intensity disturbances

5.4.1 Conversion to pasture

Converting lowland rain forest land to cattle ranches for beef production constitutes one of the main reasons for forest destruction in Central America and the Amazon (Lanly 1982). There is considerable debate on the extent and rate of pasture creation in Amazonia. Views on future developments range from a possibly continued exponential growth rate (Fearnside 1987) to a leveling off as more accessible forest tracts will become depleted (Buschbacher 1986). Since gigantic investments will be needed to provide access to ever more remote areas, the former scenario seems less likely, also because of the severe constraints imposed by soil infertility and vigorous weed growth (Alvim 1978; Buschbacher 1986).

Nevertheless, there is a very real danger that pasture creation may continue to be profitable for other reasons, e.g. land speculation (Fearnside 1987; Uhl et al. 1988b).

Generally, Amazon pastures are productive for four to eight years before they have to be abandoned. In that sense the system bears some resemblance to shifting cultivation, although the disturbance is much more severe and prolonged than the traditional clearings made by slash-and-burn cultivators (Uhl et al. 1988b).

The earliest empirical study on pasture development in the Amazon (Falesi (1976) in Buschbacher 1984) suggested that the transfer of nutrients to the soil by burning the forest could last for up to ten years. However, later work (Hecht 1982; Buschbacher 1984) concluded that the results obtained by Falesi (1976) were an exception rather than the rule and partiallly caused by an uncritical experimental design.

When studying changes in soil fertility as a function of pasture age, one can either compare sites of known age (but with possibly different initial soil fertility and management histories) or measure changes in soil nutrient concentrations over several years throughout the process of clearing etc. at one and the same site. Clearly, the latter approach is the more rigorous of the two, albeit more time consuming (cf. the discussion on direct comparisons of watersheds versus paired basin studies in section 4.3).

Buschbacher (1984) compared the nutrient dynamics and productivity, as well as total ecosystem nutrient stocks, of undisturbed mature forest and recently created pastures on Ultisols near San Carlos de Rio Negro in the Amazon Territory of Venezuela over a period of 3.5 years. The pastures were established by cutting and burning mature forest and 15-year-old secondary forest. In addition, an attempt was made to separate effects introduced by clearing per se and subsequent practices (weeding and grazing) by comparing nutrient stocks in the pastures with those in two sites that were immediately abandoned after cutting and burning.

As expected, the conversion resulted in a redistribution of nutrients from living biomass to soil and dead biomass. Cutting and burning of the mature forest produced large initial increases in topsoil available nutrient stocks (up to 50 per cent of the estimated amounts of calcium, magnesium and phosphorus present in the living above-ground vegetation before burning, and 15 per cent of that for potassium), with the rest either volatilised, leached or still held in logging debris. The pasture formed from secondary forest showed a much smaller input of nutrients to the soil and had much less unburned woody residue than that created from mature forest (Buschbacher 1984).

Topsoil nutrient levels (except total nitrogen) declined rapidly during the first year of pasture use, but changes were small in the second year, in which potassium concentrations even showed a slight increase, presumably caused by the release from decomposing logs. Nutrient concentrations at 15-30 cm depth differed little between years and were similar to those observed in undisturbed forest soil. After two years of pasture use, levels of exchangeable cations were still higher than in mature forest, whereas concentrations of available phosphorus had become similar by then. As such, the decline in soil nutrient concentrations at San Carlos proceeded at a faster rate than reported for other locations in Amazonia (Falezi 1976; Hecht 1982), a fact which Buschbacher ascribed to differences in rainfall regime, initial soil fertility (very low at San Carlos), and experimental design.

However, despite the large input of cations to the soil, Buschbacher (1984) observed an unexpected decrease in soil pH as well as a slight increase in exchangeable aluminium for both pastures, a phenomenon which he left unexplained. Since the pre-burn values of both parameters at the pasture sites were implicitly assumed to be equal to those observed in nearby undisturbed forest ("false time series"), it cannot be excluded that the pH of the "control" site was higher than that of the experimental sites before disturbance. The only way to avoid this kind of uncertainties is to adopt a pre-treatment calibration period (cf. section 4.3).

Changes in soil nutrient concentrations will be less pronounced than those in total ecosystem nutrient stocks since losses from the soil via leaching or uptake by plants will be partially compensated by inputs via decomposition of logging debris. Therefore, although soil chemical properties of the pasture land were slightly better than those of the forest some three and a half years after clearing, this was only achieved by the depletion of the nutrient reservoir built up by the forest over several centuries. As suggested by Figure 30, up to 80 per cent of the total amounts of calcium, magnesium and potassium that were originally present in the ecosystem were lost within this short time span (!).

In addition, the grassland sites derived from mature forest (PM and CBM in Figure 30) had much higher base nutrient capitals than the corresponding treatments derived from 15-year-old regrowth forest (PR and CBR) whereas the pasture sites (PM and PR) lost more bases than the immediately abandoned sites (CBM and CBR; Figure 30). The risks involved in using a "false time series" become apparent upon comparing the various stocks of organic matter, nitrogen and phosphorus. Values for the worst case (PR) are unexpectedly high (Figure 30).

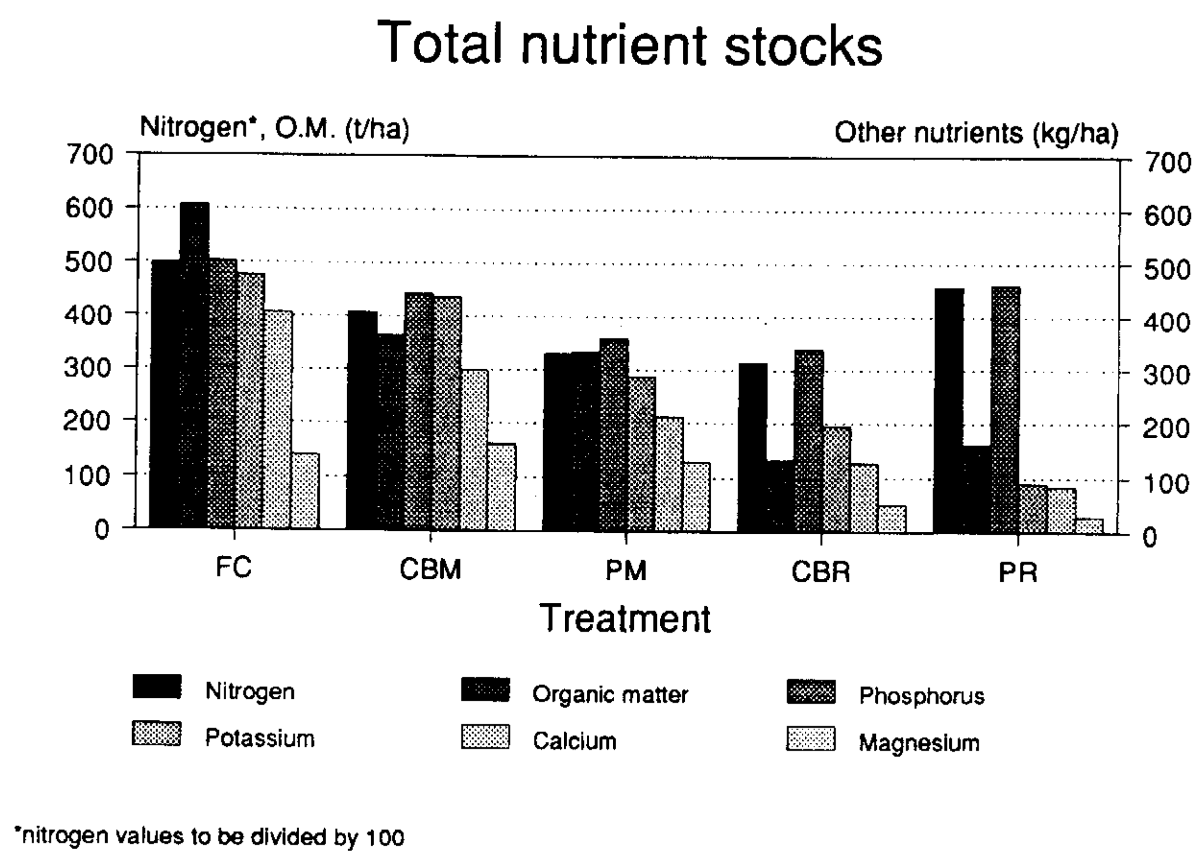

Figure 30. Total stocks of organic matter and macro-nutrients present in vegetation and soil in mature rainforest (FC), at sites 3.5 years after cutting and burning mature forest (CBM) and 15-year-old secondary forest (CBR), and in pastures formed from mature forest (PM) and from 15-year-old regrowth (PR), San Carlos de Rio Negro, Venezuela (adapted from Buschbacher 1984).

Establishment and growth of the grass (*Brachiaria decumbens*) was much more vigorous on the site formed by cutting and burning mature forest. In addition, larger amounts of nutrients were cycled in and generally fewer nutrients lost via leaching from the pasture derived from the mature forest, although such losses were considerably larger than estimated for undisturbed forest (Buschbacher 1984).

However, the validity of the quoted data is somewhat questionable in view of the way in which amounts of drainage from undisturbed forest and pastures were computed by Buschbacher (1984). In the former case (forest) these must be regarded as underestimates (cf. footnote no. 3 of Table 3), and in the latter case (pasture) as overestimates of the true values (although this may be compensated by the fact that observations of leachate quality were not initiated until six months after burning). The quoted evaporation rates for pasture land at San Carlos (30 mm mo^{-1}) seem very low indeed when compared with values reported for grassland growing under comparable climatic conditions in Guyana (85-90 mm mo^{-1}, Fritsch 1987).

Buschbacher (1984) concluded that soil nutrient stores could probably not be maintained beyond the point that all of the woody residue would be decomposed, after which the site would need to be fertilised or abandoned. Normally, Amazon pastures receive little fertilisation or other inputs to maintain their productivity (Buschbacher 1986) and are abandoned after four to eight years (Uhl et al. 1988b).

It could be argued that the severe reduction in site nutrient capital associated with conversion to grassland (Figure 30) might affect the regeneration of the forest after abandonment of the pasture. Uhl et al. (1988b) investigated regrowth rates on abandoned grassland in the eastern Amazon as a function of the intensity of previous use.

They concluded that forest regeneration was quite vigorous (ca. 10 t ha^{-1} yr^{-1}) after light use and more modest (ca. 5 t ha^{-1} yr^{-1}) in the case of moderately intense use. Conversely, regrowth on soils subjected to heavy use (including mechanical clearing of woody regrowth by bulldozer before replanting with grass) was very poor (ca. 0.6 t ha^{-1} yr^{-1}). Uhl et al. (1988b) estimated that about 10 per cent of all abandoned pasture land in their study area belonged to the latter category.

Although rates of forest regeneration were observed to be slowest on the most heavily (ab)used sites (Uhl et al. 1988b), Buschbacher et al.

(1988) reported topsoil (0-15 cm) nutrient concentrations two to eight years after abandonment to be generally independent of age and biomass of the regrowth or intensity of previous use. It was recognised that such differences were likely to exist shortly after pasture abandonment but these were thought to be masked by subsequent nutrient uptake by the vegetation and chemical reactions in the soil (e.g. phosphorus adsorption). Buschbacher et al. (1988) concluded therefore that other factors, such as soil compaction, loss of woody residue, prevention of sprouting and removal of the seed bank in the topsoil by bulldozing, were more important in explaining the slow rate of forest recovery on heavily disturbed sites than nutritional factors.

Nevertheless, the degree of nutrient depletion from the disturbed ecosystems relative to total nutrient stocks in mature forest was positively correlated with degree of intensity of pasture use (cf. Figure 30). Unfortunately, despite the apparent recuperative potential of many abandoned Amazon grasslands, secondary forests in the region become increasingly often disturbed by fire (Uhl & Buschbacher 1985).

5.4.2 Conversion to forest plantations and extractive tree crops

The area of forest plantations in the tropics almost tripled between 1965 and 1980 and the rate of planting in the 1980's was expected to double that of the 1970's (Evans 1982). Recently, concern has been expressed about the possible negative effects that fast-growing tree plantations may have upon soil nutrient reserves, especially when grown in short rotations on poor tropical soils (Chijioke 1980; Hase & Fölster 1983; Jorgensen & Wells 1986; Ruhiyat 1989).

As shown in the previous sections, depletion of soil nutrient reserves during the cropping phase of shifting cultivation or after conversion to pastures manifests itself in a more or less rapid decline in crop yields. However, when a natural forest is replaced by a tree plantation, the evaluation of a possible future decrease in productivity (the "second-rotation problem") becomes a far more complex affair.

Firstly, the determination of nutrient losses in harvested produce is a laborious task and surrounded with practical problems at every stage (Auchmoody & Grewelin 1979). Furthermore, the quantification of nutrient inputs (via bulk precipitation and rock weathering) and losses (soil erosion, deep leaching) requires the long-term collection of climatic and hydrological data as well as the analysis of numerous water samples (Likens et al. 1977; cf. chapter 3). Thirdly, there is the long time span inherent to forest production.

An alternative approach, therefore, involves the comparison of soil- or ecosystem nutrient stores in a "false time series" (pre-treatment vs. post-treatment situation; Hase & Fölster 1983, Buschbacher 1984). However, as indicated earlier, this method suffers from the (severe) limitation that temporal changes may be confounded by spatial variability in soil characteristics. In addition, it is rather insensitive (Verstraten 1980) and only major differences between sites can be detected in this way.

Whatever the approach adopted, there is the additional difficulty of interpreting the results. This is caused by the fact that the soil nutrient store cannot be defined clearly in forestry. As pointed out by Hase & Fölster (1983), the determination of amounts of available nutrients in forest soils by chemical extraction methods still lacks verification by bioassay techniques. Also, the lower boundary of the forest root network is often poorly defined and it becomes difficult to establish to what extent nutrients that are released by weathering will be available for uptake by the trees or leached from the system (chapter 3).

Information on nutrient immobilisation in tropical tree plantations is gradually becoming available. Examples are the studies on *Pinus caribaea* and *Gmelina arborea* in Nigeria and Brazil (Egunjobi & Bada 1979; Chijioke 1980; Russell 1983), on *Tectona grandis* in Nigeria (Nwoboshi 1983) and Venezuela (Hase & Fölster 1983), on *P. patula* and *Cupressus lusitanica* in Tanzania (Lundgren 1978), on *P. merkusii* and *Agathis dammara* in Indonesia (Bruijnzeel 1984; Bruijnzeel & Wiersum 1985), and on *Eucalyptus spp.* in Brazil (Bellote et al. 1980; Poggiani et al. 1983a,b), southern India (George 1984) and East Kalimantan, Indonesia (Ruhiyat 1989).

Biomass production of fast-growing hardwood species such as *Gmelina* or eucalypts may (Jorgensen & Wells 1986) or may not (Russell 1983; Bruijnzeel 1984) exceed that of tropical conifers. However, amounts of nutrients incorporated in these hardwood plantations are normally higher than those in softwood plantations of similar age (Russell 1983; Bruijnzeel 1984), mainly as a result of the higher nutrient concentrations in hardwood tissue (Jorgensen & Wells 1986). As such, demands on soil nutrient reserves will differ between species.

In contrast to the growing body of information on tropical tree biomass and nutrient content (and hence on potential nutrient losses upon harvesting) and on nutrient inputs from the atmosphere (Bruijnzeel 1989a), there is a real paucity of reliable information regarding hydrological losses of nutrients from tropical forests and therefore on rates of chemical weathering (Clayton 1979; chapter 3).

Even less is known with respect to the critical early phases of plantation establishment (clearing, burning, planting) during which period both volumes and nutrient concentrations of water percolating through the soil are enhanced (Russell 1983; cf. Table 4.1 and Figures 18 and 27) and losses via leaching and soil erosion are likely to be maximal.

Studies of this kind are now in progress in Indonesia and Sabah and a third one has been initiated recently in Peninsular Malaysia (Abdul Rahim personal communication). Meanwhile, it is unfortunate that the only two studies for which results on changes in water chemistry during the conversion of tropical forest to plantations are available (i.e. Jari Florestal, Brazil (Russell 1983) and Sungai Tekam, Peninsular Malaysia (DID 1986, 1989)) both suffer from methodological shortcomings.

The Brazilian study employed rather arbitrary (although not entirely unreasonable) methods to derive (not measure) amounts of drainage associated with various stages of plantation development in a "false time series", whilst results for Sungai Tekam were based on insufficiently frequent water sampling (Zulkifli Yusop 1989). As expected, both studies reported temporarily increased nutrient concentrations in soil- or streamwater (Russell 1983; DID 1986, 1989; cf. Buschbacher 1984; Uhl & Jordan 1984).

In the absence of reliable estimates of dissolved nutrient exports after clearcutting tropical forest, it is worthwhile to consider results obtained in warm-temperate areas such as the southeastern USA (Fischer 1981; Riekerk 1983; Hewlett et al. 1984; Van Lear et al. 1985) and eastern Australia (Hopmans et al. 1987).

According to Hewlett et al. (1984) a "typical southern clear-cut operation" in mixed secondary forest on Ultisols/ ultic Alfisols produced only minor and short-lived increases in dissolved ion exports. These were essentially caused by the increase in water yield following the clearcut, since concentrations of individual elements either did not respond to the treatment or decreased because of a dilution effect. Slash was not burned but roller-chopped before the area was planted mechanically with loblolly pine (Hewlett et al. 1984).

Repeated pre-harvest low-intensity prescribed fires to control understory vegetation at a comparable site in South Carolina had no effect on water quality either (Van Lear et al. 1985).

Clearcutting followed by roller-chopping and replanting of natural pine forest on poorly drained sandy ultisols in coastal Florida resulted in a similarly modest effect on hydrologic nutrient exports (Riekerk 1983). Effects on both streamflow and nutrient losses during the first year after the treatment were roughly two to three times greater when cutting was followed by mechanical stump removal and burning/windrowing of slash (cf. Kang & Lal 1981). Effects were already negligible during the second year after clearing (Riekerk 1983). It should be added, however, that the two-year post-treatment period was noticeably dry, whilst the initial calibration period (one year) had been quite wet (Riekerk 1983). Unfortunately, any effects of these climatic irregularities on the comparison of pre- and post-treatment hydrologic nutrient losses were not addressed properly in this study. In both treatments, however, the extra losses were less than annual nutrient inputs from atmospheric sources (Riekerk 1983). In addition, they were much smaller than nutrient losses associated with harvested produce and burning (Fisher 1981).

Equally small dissolved ion exports were reported by Hopmans et al. (1987) following clearcutting and burning of eucalypt forest in south-eastern Australia. Apart from somewhat increased potassium concentrations, increases in hydrologic nutrient losses during the first year after the operation were entirely due to the increase in water yield. Hopmans et al. (1987) ascribed this lack of response to the presence of a thirty-metre wide buffer strip which had been maintained on either side of the stream channel. Again, hydrologic nutrient losses were a mere fraction of the losses due to burning the original forest or of the amounts incorporated in the fast-growing pines that replaced the eucalypts (Stewart et al. 1981; Stewart & Flinn 1985). Inputs of calcium via rock weathering (as computed via the small catchment mass balance method;

Clayton 1979; Bruijnzeel 1983b) were low and might cause productivity problems during future rotations (Hopmans et al. 1987).

It would be premature, however, to conclude from the above examples that leaching losses associated with clearcuts in the humid tropics would also be small. For one thing, the rainfall surplus over evapotranspiration is considerably larger in the tropics (chapter 2) and with it the potential intensity of leaching.

In addition, due to the large standing biomass of mature tropical forest, considerably larger amounts of nutrients may be removed in harvested stems and burning of logging debris (Hase & Fölster 1983; Russell 1983; Stewart and Flinn 1985). On the other hand, rates of regeneration and nutrient uptake will be faster in the tropics (Uhl & Jordan 1984). Clearly, there is a need for more quantitative information in this respect and it makes one look forward to the results of the Malaysian and Indonesian studies mentioned previously.

No single study has managed to adequately quantify all gains and losses of nutrients for a complete cycle of tropical forest clearing, followed by plantation establishment, maturing and harvesting. However, approximate nutrient budgets for entire forest rotations have been computed for teak plantations on eutrophic soils in Venezuela (Hase 1981; Hase & Fölster 1983), for fast-growing conifers on fairly fertile volcanic soils in Indonesia (Bruijnzeel 1984; Bruijnzeel & Wiersum 1985) and for fast-growing hard- and softwoods on oligotrophic soils in Amazonia (Russell 1983; 1987). These are discussed briefly in the following.

In western Venezuela sites with light textured soils ("Banco" sites with Inceptisols) are considered suitable for reforestation with teak. The natural vegetation of these sites is a semi-evergreen seasonal forest (average rainfall 1800 mm yr^{-1} with a pronounced dry season), which may attain an above-ground biomass figure of about 400 t ha^{-1} (Hase & Fölster 1982). Hase & Fölster (1983) presented data on nutrient stocks in vegetation and soils (0-100 cm) of ten teak plantations ranging in age between six months and nine years as well as in mature forest of the type that preceded the plantations.

In addition, they measured atmospheric nutrient inputs for one year and derived contributions by weathering from the observation that ca.

five per cent of the primary silicate minerals present in the sand and silt fractions of the soils had been transformed into kaolinite clays, combining this finding with an estimated age of the top 100 cm of the soil of about 1000 years (Hase 1981).

A tentative nutrient budget for the first eighty years after forest clearing and planting teak was constructed on the basis of the above original data in combination with data on timber yield of teak plantations elsewhere in the tropics, stemwood nutrient contents of 33- and 38-year-old teak trees in northern India, and averaged leaching losses for a number of temperate and tropical forest ecosystems (Hase & Fölster 1983). Erosion during plantation establishment or at a later stage (Bell 1973) was not taken into consideration. Table 7 summarises the results.

Considerable amounts of nutrients appear to be lost from the site with the removal of merchantable timber upon clearing the natural forest and as a result of increased leaching during the first year (Hase 1981). Still larger amounts are removed in harvested teak boles at the end of the first rotation. Total nutrient losses are in all cases significantly higher than additions via precipitation and weathering, thereby depleting soil nutrient reserves.

As shown in Table 7, by far the most serious loss associated with the conversion to teak and the subsequent harvest of logs is that of calcium. Although these estimates must be considered as orders of magnitude only, because of the uncertainty introduced by the liberal use of data from the literature, it is clear that calcium will be the prime element limiting future productivity, possibly so already during the second rotation (Hase & Fölster 1983).

A rough check of the latter assertion can be obtained by assuming similar nutrient inputs and losses (harvest of teak boles and "steady state" leaching only) for the second rotation and comparing the net losses so obtained with approximate nutrient stocks present at the start of the latter (see footnotes Table 7). Indeed, estimates of calcium exports associated with the second rotation vary between 90 and 395 per cent of initial stocks (Table 7). Most of this variation is due to the variability of calcium concentrations in stemwood of mature teak trees as reported in the literature (Hase & Fölster 1983). However, similarly high concentrations of calcium and magnesium were found in the stems of a 25-year-old teak plantation on Inceptisols in Indonesia, whilst values for

TABLE 7. Approximate nutrient budget (kg ha^{-1})[1] for the first rotation of teak production (80 years) on loamy "Banco" soils in western Venezuela (adapted from Hase & Fölster 1983)

	P	K	Ca	Mg
Inputs				
Rainfall	40	475	580	165
Weathering	?	425-535	50-110	105-135
Total input	>40	900-1010	630-690	270-300
Outputs				
Harvest of mature forest	25	430	980	50
Extra leaching	?	0-700	?	?
Steady leaching	10	185	480	170
Harvest of teak boles	490	805	2935-4680	545
Total losses	525	1770	4395-6140	765
Net losses	485	760-870	3705-5510	465-495
Total stock in natural forest	4890[2]	2290	6725	1760
Net loss as % of total stock	10[3]	33-38	55-82	26-28
Approximate remaining stock at start of 2nd rotation[4]	4400	1215-1535	1145-3025	1180-1300
Inputs	>40	900-1010	630-690	270-300
Outputs	500	990	3415-5160	715
Net losses	460	-20-90	2725-4530	415-445
Net loss as % of stock at start of rotation	10[5]	0-7	90-395	32-38

1 figures rounded off to nearest 5
2 of which ca. 2300 kg ha^{-1} as available phosphorus in the soil
3 21 per cent of available phosphorus reserves
4 excluding slash from teak harvest
5 25 per cent of available phosphorus reserves

phosphorus and potassium were 19 and 194 per cent higher respectively than the ones on which the computations of Table 7 are based (Bruijnzeel 1983a). Interestingly, rates of soil nutrient depletion were much lower for the less fertile sandier sites, reflecting the lower productivity (and rate of nutrient immobilisation) of these sites (Hase 1981).

Since the observations of Hase & Fölster took place in 1977/78, it would be interesting to see whether repeated sampling in the oldest teak stands (which would be about twenty years old now) would confirm the trends predicted in Table 7. Also, since the hydrology of the natural forest in the area is well researched (Franco 1979), additional observations of soil- and streamwater composition would improve estimates of amounts of nutrients supplied by weathering and lost via leaching. However, as pointed out by Hase & Fölster (1983), most of the uncertainty surrounding the long-term nutrient dynamics of these teak plantations is related to stemwood nutrient contents of mature trees and it is here that further information is needed most.

Bruijnzeel & Wiersum (1985) presented a nutrient balance sheet for ideally stocked plantations of *Agathis dammara* grown in rotations of 40 years on Andosols in Java. Atmospheric nutrient inputs, contributions by weathering, and deep leaching were determined via the small-watershed mass balance method (Bruijnzeel 1983b). Losses in harvested produce were based on measurements of biomass and nutrient contents in a "false time series" of four plantations ranging in age between seven and 35 years and growing on the same soil type (Bruijnzeel 1983a).

Plantation forestry in Java is usually practised on steep slopes and employs the "taungya" system (King 1968) during the early stages of plantation establishment. As such, the soil is exposed for about three years, during which period soil erosion may occur. Bruijnzeel & Wiersum (1985) estimated nutrient losses associated with erosion during taungya from pre-war erosion studies in Java (Coster 1938; Gonggrijp 1941a) and made a distinction between "moderate" (1 cm yr^{-1}) and "heavy" (2.5 cm yr^{-1}) erosion. Recent work on similar soils in West and East Java by Bons (1990) and Rijsdijk & Bruijnzeel (1990) respectively suggested erosion rates of 35-70 t ha^{-1} yr^{-1} (ca. 0.5-1.1 cm yr^{-1}) during taungya, the bulk of which was produced by those parts of the fields that were used as access routes (e.g. field boundaries; cf. section 4.5.1). No estimates of extra losses due to increased leaching during taungya were made at the time (Bruijnzeel & Wiersum 1985).

Their results are summarised in Table 8, supplemented with an estimate of initial leaching losses (see footnotes). To facilitate comparison with Table 7, nutrient losses associated with trunk removal (including bark) are given, even though the prevailing practise generally involves on-site debarking. Since branches are usually collected for use as fuelwood and (in certain parts of Java) *Agathis* leaves are also used for fodder/compost making, several harvesting regimes have been included in Table 8.

A comparison of total gains and losses of nutrients over the rotation shows that inputs are sufficient to allow the regular harvesting of bole wood (incl. bark) plus branches, but that total tree harvesting is most likely to deplete soil reserves to an unacceptable level, especially so for phosphorus (Table 8).

However, it is important to distinguish between fractions of soil phosphorus that are readily available (ortho-P in soil solution), non-available (locked up in (an)organic compounds) and potentially available (organic P) (Sanchez 1976). The methods employed by Bruijnzeel & Wiersum (1985) extracted readily available and total phosphorus but not organic phosphorus. As such, their estimate of available phosphorus will be highly conservative. Total amounts present in the soil amounted to 5-6000 kg ha^{-1} (Bruijnzeel 1983a). Van Barneveld et al. (1984) reported concentrations of available phosphorus (determined by the Bray-II technique which extracts larger amounts than the relatively mild citrate method) for a *damar* plantation in East Java underlain by a soil that was morphologically and chemically very similar to the soils encountered by Bruijnzeel & Wiersum, which were up to two times higher. However, even if the available phosphorus stock were double the presently quoted value, total tree harvesting would still exhaust available phosphorus reserves after one full rotation. Further work on the availability of phosphorus to trees in these volcanic soils is necessary and intended.

Bruijnzeel (1984) conducted a similar analysis for *Pinus merkusii* plantations in the vicinity of the above *Agathis* stands grown in rotations of 25 years and on more fertile soils. Nutrient uptake rates over 25 years by the (overstocked) pine stands were considerably lower than those for *Agathis* and in fact enabled total tree harvesting without any serious decline in soil fertility.

TABLE 8. Nutrient budget (kg ha^{-1})[1] for an ideally stocked plantation of *Agathis dammara* under various management conditions and a rotation period of 40 years (adapted from Bruijnzeel & Wiersum 1985)

	P	K	Ca	Mg
Inputs				
Rainfall	35	385	395	160
Weathering	200	1575	3060	1565
Total inputs	235	1960	3455	1725
Outputs				
Leaching	30	880	1160	1220
Extra leaching during taungya[2]	2	25	15	15
Erosion[3]	2	15-25	20-140	10-40
Harvest of crops[4]	20	40	20	7.5
Harvest of stemwood only	105	435	660	155
Idem + bark	145	560	1295	250
Idem + branch	205	855	1680	330
Total tree[5]	350	1655	3315	705
Losses / Gains				
Bole harvest	0.85	0.78	0.74	0.88
Bole + branch	1.10	0.93	0.86	0.92
Total tree	1.72	1.34	1.33	1.14
Nutrients in soil (0-100 cm)[6]	90-110	540-1340	750-4130	340-1220
Net loss as % of reserves[7]	155-189	49-122	25-151	20-72

1 figures rounded off to nearest 5

2 assuming an increase in water yield equal to rainfall interception in a mature plantation (ca. 670 mm (Bruijnzeel 1988; cf. Table 4; a doubling of potassium concentrations and no changes in streamwater concentrations of phosphorus, calcium, and magnesium (C.A. Bons, personal communication)

3 assuming a rate of 1 cm yr^{-1} for three years of taungya; available nutrients only

4 upland rice and maize, grain only as straw is usually burned on-site (based on concentrations in Ochse et al. 1961)

5 not including undergrowth

6 readily available: extractable by NH_4-acetate at pH 7 (Ca, Mg, K) or a 2% solution of citric acid (P)

7 in case of total tree harvest

The largest single conversion of tropical rain forest to plantations is situated at Jari Florestal in eastern Amazonia. The plantations (mainly *Pinus caribaea* and *Gmelina arborea*) are mostly on infertile sandy Oxisols and Ultisols (Russell 1983).

Russell (1983) determined forest biomass and total ecosystem nutrient stocks for a "false time series" consisting of (1) undisturbed natural forest, (2) six-month-old pines, (3) an 8.5-year-old *Gmelina* stand, (4) a 9.5-year-old pine stand, and (5) a six-month-old pine plantation on a site where *Gmelina* had been grown for 8.5 years. All sites were on sandy Ultisols with very similar physical characteristics, on the basis of which Russell assumed their original chemical properties to have been the same as well. As such, all differences in soil nutrient reserves between sites were ascribed to the various treatments whilst possible differences in initial site fertility were ignored.

The hazards of this approach become readily clear from the data presented by Russell (1983) on nutrient stocks in natural forest and mature plantations. For example, total stocks of phosphorus and calcium were larger for the *Gmelina* site than in the natural forest. Although these differences are probably not statistically significant due to the large standard errors of the biomass estimates, it cannot be excluded that this finding reflects a real difference in original site quality. Also, soil reserves of calcium and magnesium in the mature pine stand amounted to only 13 and 18 per cent respectively of those in the *Gmelina* plantation whereas amounts of the two elements incorporated in tree biomass differed by only 20 per cent between species (with the higher values for the slightly older pines).

Of course, one could think of a variety of factors to explain such differences between sites, ranging from differences in original forest biomass or amounts of slash left upon clearing, to contrasting site fertility. Whatever the true reason, however, it will be clear that there can be no substitute for recurrent inventories of soil chemical characteristics at the same location (Chijioke 1980).

In addition to the determination of total system nutrient stocks, Russell (1983) collected data on atmospheric nutrient inputs for one year and on the chemical composition of free-draining soil water in the various treatments. By combining the information on soil water quality with (theoretical) estimates of annual drainage rates an idea was obtained of nutrient leaching losses associated with the various phases of the time series (cf. the discussion in chapter 3). In view of the above difficulties with the data set for *Gmelina*, the following discussion will concentrate on the conversion of natural forest to pine plantations only, and avoid a comparison of total nutrient stocks for different sites (Table 9).

Net losses of nutrients associated with the first cycle of forest production were considerable, removing 62 (phosphorus) to 83 (magnesium) per cent of the total nutrient store present before the conversion. Up to 45 per cent of (gross) losses consisted of nutrients removed by the harvesting of timber from the natural forest whereas only a modest proportion (11-17 per cent) was associated with the harvest of pine stems (with the exception of phosphorus). Leaching losses of calcium, magnesium and potassium appeared sizeable as well, although it should be remembered that these estimates may not be too reliable.

Since extra leaching losses during the establishment of the second rotation were small (Russell 1983), net losses associated with the second cycle were only a fraction (12-21 per cent, again with the exception of phosphorus: 54 per cent) of those determined for the first period (Table 9). Both on the basis of the total nutrient inventory for the mature pine stand given by Russell (1983) and the inferred remaining stocks after the first rotation implied by Table 9, it would seem as if magnesium would be the prime element limiting future forest productivity, followed by phosphorus, calcium and potassium.

The number of rotations after completion of the first cycle that would be possible before available nutrient stocks would be exhausted, would be less than two (magnesium and phosphorus), two to three (calcium), and less than four (potassium). Similar figures were derived by Russell (1983), with the notable exception of potassium for which he computed 16.7 rotations before complete exhaustion. Since the latter value was based on a comparison of nutrient inventories of contrasting sites, the presently derived value of 3.7 may be more realistic, also in light of the values obtained for the other elements (Table 9).

It must be concluded, therefore, that (pine) plantation forestry at Jari, and presumably also at other locations experiencing similar soil conditions, is not sustainable without extra inputs from fertilisers (Table 9).

TABLE 9. Approximate nutrient budget (kg ha^{-1})[1] for first and second rotations of pine production (10 years each) on sandy Ultisols, Jari Florestal, Brazil (adapted from Russell 1983)

	P	K	Ca	Mg
Inputs				
Rainfall[2]	1.5	100	160	35
Weathering[3]	-	-	-	-
Total input	>1.5	>100	>160	>35
Outputs				
Harvest of mature forest	30	375	520	175
Leaching[4]	1	445	480	365
Pine harvest[5]	35	100	205	85
Total losses	66	920	1205	625
Net losses	65	820	1045	590
Total stock in natural forest	105	1055	1420	710
Net loss as % of total stock	62	78	74	83
Net losses during second rotation[6]	35	95	215	110
Net loss as % of remaining stock[7]	59	27	38	63
No. of rotations after 1st before exhaustion[8]	1.7	3.7	2.6	1.6

1 figures rounded off to nearest 5
2 underestimate since particulate matter not included in samples
3 considered negligible by Russell (1983)
4 adding losses estimated by Russell for first year after logging and burning to those for young pine plantation ("second" year) and interpolating between the latter and value for mature pine to obtain losses for intermediate years; no extra losses assumed for volatilisation
5 at age ten, values extrapolated from data for 9.5-yr-old stand
6 assuming only steady state leaching losses (Russell 1983) and pine bole hearvest
7 computed as initial stock minus losses associated with first rotation and adding nutrients contained in branches and foliage of mature pines
8 remaining stocks at end of first cycle divided by net loss of second (and presumably subsequent) cycles

Fertilisers are of course already widely used in extractive tree plantations such as rubber or oil palm (Thajib & Pushparadjah 1984). An extra problem with the evaluation of nutrient balances in the case of fertilised plantations consists of increased leaching losses (see Agamuthu & Broughton 1982; Pushparadjah 1982; Foong et al. 1983; Chang & Chow 1985). However, an evaluation of the economic feasibility of fertiliser use in non-extractive tropical tree plantations (cf. Evans 1982) is beyond the scope of the present analysis.

5.4.3 Conversion to permanent agricultural cropping

Naturally, much of what has been said about the initial effects of forest clearing and conversion to pasture land or tree plantations also holds for the use of tropical forest land for permanent agriculture. In addition, it will be clear from the foregoing that if low-intensity tree farming and the exploitation of pastures on poor soils are not sustainable without the use of fertilisers, neither will be intensive agriculture.

However, rather than investigating the feasibility of applying fertilisers to maintain the productivity of inherently poor soils in remote tropical areas (Sanchez et al. 1982; 1983), it is of interest to examine whether the nutrient dynamics of the various forms of land use which combine the growing of woody perennials with annual cropping (often grouped under the general term "agroforestry") are necessarily any better than those associated with "common" agricultural systems (Figure 31; Plate 16).

There can be no doubt that trees exert a number of positive influences on both physical and chemical soil properties, some of which are well-documented and experimentally demonstrated, others still largely unproved (Young 1986). As such, according to some, the introduction of trees in agricultural cropping systems might mitigate some of the negative effects of continuous cropping (cf. Figure 31). However, as pointed out by Wiersum (1988), the impact of tree planting on agricultural fields will vary greatly with local conditions (e.g. slope steepness, fertility of substrate, rainfall regime, etc.) and agroforestry should not be seen as a panacea for all soil management problems in the tropics.

One of the main principles of soil management in agroforestry is to make the best use of its resource conserving and -sharing potentials

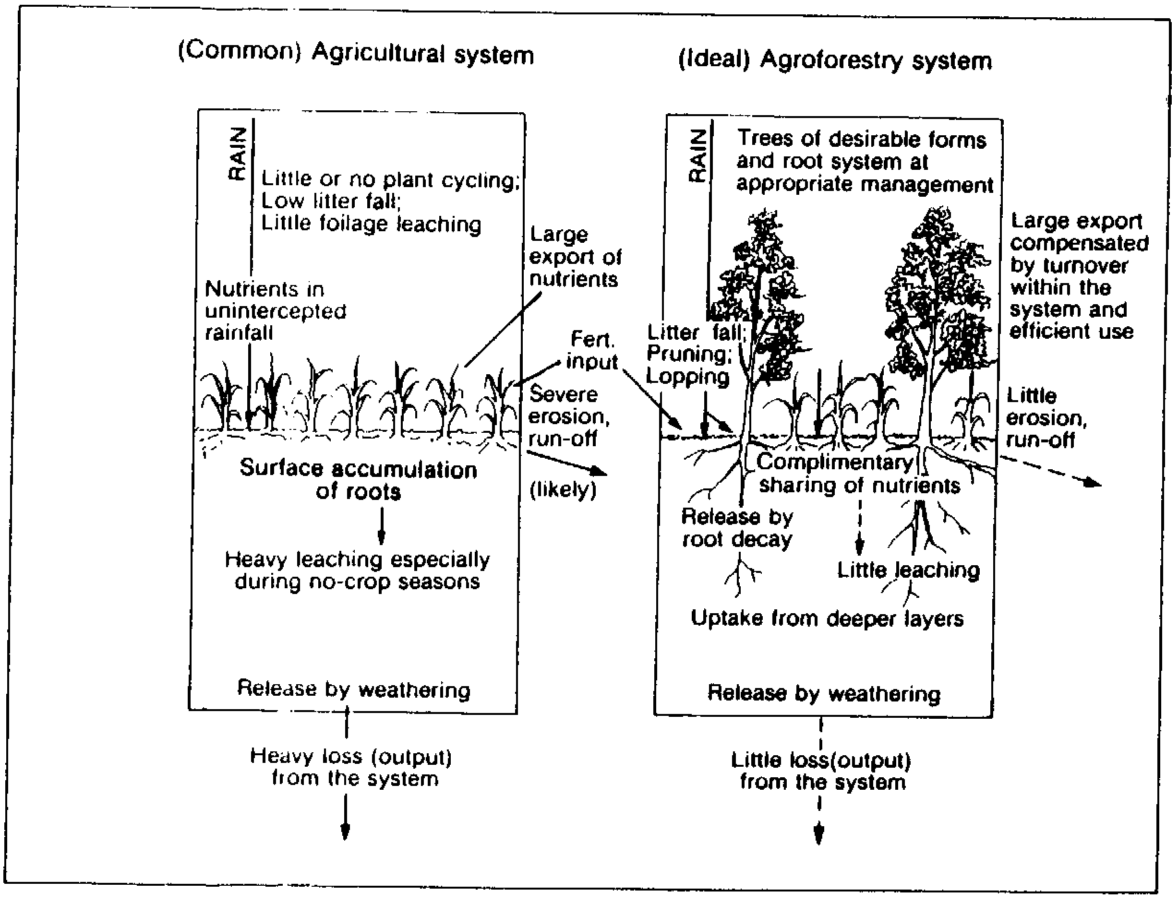

Figure 31. Schematic representation of nutrient relations and perceived advantages of an ideal agroforestry system in comparison with a "normal" agricultural system (adapted from Nair 1984).

Nair 1984). An example of the former is the planting of nitrogen fixing trees (e.g. *Leucaena leucocephala*) in contour hedgerows behind which eroded soil may accumulate and form "natural" terraces. In addition, such hedgerows may supply nitrogen-rich prunings that can be used as mulching material between the annual crops, or as fuelwood or staking material (Kang et al. 1981). The latter authors showed how such an "alley cropping" system was able to sustain reasonable grain yields on a nitrogen deficient sandy soil in southern Nigeria.

The overriding importance of a well-developed litter- or mulch layer in checking nutrient losses via soil erosion in agroforestry systems was reviewed by Wiersum (1984)(see also Lal (1983,1990). However, in some cases it may be a problem to obtain mulching material in sufficient quantities to be effective (Nair 1984).

An example of resource-sharing is the use of leguminous shade trees in coffee and cocoa plantations. Aranguren et al. (1982ab) demonstrated that amounts of nitrogen in litterfall form the shade trees in unfertilised plantations were more than five times those removed in harvested produce (coffee beans, cocoa pods, shells returned to the fields after processing). However, they added that studies of leaching and denitrification rates were needed to substantiate their preliminary conclusion that inputs of nitrogen by the shade trees were sufficient for sustainable production.

Only recently fairly complete studies have become available for cocoa plantations under different types of shade trees (Alpizar et al. 1986; Heuveldop et al. 1988; Fassbender et al. 1988; Imbach et al. 1989) and to a lesser extent for coffee plantations, both in Costa Rica (Fassbender 1987; Fassbender & Alpizar 1987).

The Costa Rican work on nutrient dynamics of cocoa plantations confirmed the results that Aranguren et al. (1982a) had obtained for nitrogen. Nutrient losses in harvested produce were again much less than corresponding amounts in litterfall or immobilised in the vegetation (Heuveldop et al. 1988). Since amounts of potassium removed in beans and husks and those taken up by the vegetation were quite substantial compared to available reserves in the soil (and in fact exceeding fertiliser additions: Fassbender et al. 1988), it might be worthwhile to consider returning the husks (containing 75 per cent of "harvested" potassium) to the fields after processing and/or composting (cf. Aranguren et al. 1982a).

Interestingly, concentrations of total nitrogen in soil water (0-30 cm depth) were 2-2.5 times higher under leguminous shade trees (*Erythrina*) compared with laurel shade trees (Fassbender et al. 1988). However, soil water at one metre depth did not show any such difference (Imbach et al. 1989), suggesting that the extra nitrogen fixed had been taken up already by the root system. In contrast, there were marked differences in calcium, magnesium, and (to a lesser extent) potassium concentrations in subsoil water, with the highest concentrations invariably occurring under the system employing leguminous shade trees (Imbach et al. 1989). Since the water balances (as computed by somewhat insensitive methods) were almost identical for the two plantations, amounts of bases leached from the two systems reflected the differences in water quality (Imbach et al. 1989).

A comparison of atmospheric nutrient inputs with (derived) leaching losses suggested accumulations of nitrogen, phosphorus and potassium, whereas calcium and magnesium were lost from the ecosystem (Imbach et al. 1989). Although this finding could also be partially explained by the fact that the study year was a rather dry one (limited percolation) and by the relatively high nutrient inputs determined by these authors (generally 2-5 times those observed by Hendry et al. (1984) for the same location), Imbach et al. (1989) concluded that the nutrient budgets for the two agroforestry systems were more favourable than many others, particularly annual cropping systems.

However, since leaching losses from a similar plantation on a more fertile Alfisol in eastern Brazil were much higher (Leite 1985), it would be premature to ascribe the favourable nutrient balances observed in the examples from Costa Rica entirely to the use of a mixed cropping system.

Agroforestry systems clearly have a number of advantages over annual cropping systems (or indeed shifting cultivation in densely populated areas: cf. Nair (1984); Ewel (1986); Young (1986); Vandermeer (1989). As such, the results obtained to date do hold some promise. In view of the large amount of work that is still to be done with respect to the water and nutrient dynamics of various agroforestry systems, it makes one look forward to the results that the recently started Tropical Soil Biology and Fertility (TSBF) programme of the International Union of Biological Sciences (Swift 1985) may produce.

6 SUMMARY AND CONCLUSIONS

(1) Although estimates of the rate and extent of deforestation (here defined as the net conversion from forest to non-forest vegetative cover) in the humid tropics differ considerably between specialists there is at least general agreement that the problem is taking on serious proportions requiring immediate action. At the same time there is an increasing tendency to believe that large-scale planting of trees will solve most, if not all, problems. Before policy makers can take effective action in this respect they need to be properly informed on the real impacts of various (de)forestation activities. The present paper, aimed initially at the scientific community, critically reviews the literature on water and nutrient budgets of undisturbed tropical forests and effects of disturbance or conversion to other land use types.

(2) There is considerable diversity in hillslope hydrological behaviour (storm runoff generation) of undisturbed moist tropical forest areas, which mainly relates to variations in geological (and therefore geomorphological and pedological) settings. Infiltration excess overland flow (HOF) is a fairly rare phenomenon under undisturbed conditions but widespread (hillside) saturation overland flow (SOF) has been reported in several instances where soil permeability decreased rapidly with depth. In the case of deep and permeable soils and straight or convex slopes, throughflow is the dominant supplier of storm flow (cf. Figure 5).

(3) Annual evapotranspiration (ET) for tropical lowland forests that rarely experience serious soil moisture shortages (as determined by the water balance method mainly) averages about 1415 mm (range 1310 - 1500; n = 11; Table 1); this value may fall to 900 mm in the case of seasonal forest. Catchment leakage presented a problem in many cases.

(4) Corresponding values for montane forests (excluding "cloud forests") converge at about 1225 (range 1155 - 1295) mm yr-1 (n = 5; Table 2), with no strong trends with altitude or annual precip-

itation. Basin leakage may have influenced the above value, however. There is a need for more hydrological studies of montane tropical forest.

(5) "Cloud forests" (*sensu* Stadtmuller 1987) represent a special case, showing very low ET rates as a result of low radiation inputs, low atmospheric vapour pressure deficits and the process of "cloud stripping"; available data suggest "gross" (i.e. including contributions by cloud stripping) values for ET of 310 - 390 mm yr-1 (Table 2).

(6) Of the two main components of ET, rainfall interception (Ei) has frequently been overestimated because of inadequate sampling designs for measuring net precipitation (i.e. throughfall plus stemflow). Relatively large numbers of gauges that are re-located at regular time intervals at random locations on the forest floor are required for reliable estimates.

(7) The best studies of net precipitation in tropical forests suggest average values of 86 (range 77 - 93) per cent of incident precipitation (on an annual basis) for lowland forests (n = 13) and of 82 (range 75 - 86) per cent for (lower) montane forests (n = 6). Since stemflow usually constitutes 0.5 - 2 per cent of rainfall, average Ei values for the two groups of forests amount to ca. 13 (range 4.5 - 22) and 18 (range 10 - 24) per cent of incident precipitation respectively. Annual Ei in "cloud forest" ranges from ca. 10 per cent to negative values due to cloud stripping, particularly on exposed locations.

(8) The second major component of ET, transpiration (Et), is often only known indirectly (Et = ET - Ei) and unreliably. Values so determined converge around 1045 (range 885 - 1285) mm yr-1 (n = 9) for lowland forests never severely short of water to about 600 mm yr-1 for (semi)deciduous forest; estimates for lower montane forests (excluding "cloud forests") do not correlate with site elevation and vary considerably (510 - 830 mm yr-1); the available estimates for "cloud forest" range from 285 to 510 mm yr-1 (values corrected for cloud

stripping). More work is needed on Et of tropical forest, particularly on stomatal behaviour and in montane forests.

(9) Atmospheric inputs and hydrologic outputs of calcium, magnesium, potassium, phosphorus and nitrogen for 25 (23 published and two constructed by the present writer) (sub)tropical forest ecosystems on a variety of geological substrates were evaluated critically in terms of methodology. The data set was subdivided into lowland (n=19) and montane sites (n=6). The lowland group was further subdivided into forests on moderately to very infertile soils (n=12), and on relatively fertile soils (n=7; Table 3).

(10) Variation in reported nutrient fluxes, not only per element and fertility group, but also per lithology and even for particular sites (depending on methodology and/or investigator) is large and partly reflected the variable quality of the data. Much of the variation seems related to enrichment/contamination of precipitation samples by regional or local dust, fire or organic debris, sometimes coupled with sub-optimal analytical facilities; improper estimation of amounts of drainage/streamflow is another major factor. Correction of the underlying hydrological assumptions often transformed an apparently positive budget (i.e. nutrient accumulation) into a negative one (net nutrient loss). Discrepancies between estimates based on lysimetry and streamflow sampling are especially large for forests on soils of low fertility.

(11) Scatter plots of annual runoff vs. annual calcium, magnesium and potassium losses for 19 sites (mostly small catchments) reveal four groups with characteristic nutrient export patterns. Calcium and magnesium losses from Spodosols and highly leached Oxisols are distinctly lower than for the overall Ultisol/Oxisol group, whereas losses from sites with Inceptisols exceed those from the latter group. Still higher losses have been recorded for sites underlain by rocks that were particularly rich in these elements, such as limestones, calcareous shales and slates. Potassium exports show a similarly upward trend over the first three groups, with some overlap between groups. Potassium losses from the fourth group are low,

reflecting the scarcity of the element in the underlying bedrock. Montane forests did not exhibit any special patterns with respect to these three elements.

(12) Phosphorus is accumulating in virtually all cases, reflecting the low mobility of the element (immobilisation by ferro-aluminium compounds in the soil). Nitrogen budgets are only partially covered by water-bound pathways. A full evaluation would need to take biological fixation inputs and denitrification outputs into account as well.

(13 Losses of metal cations from areas with highly infertile soils are quite high when determined by sampling streams draining catchments of intermediate to large sizes. Losses from such areas are much smaller when evaluated on the basis of small catchment areas and closer to the losses actually experienced by the vegetation. Such discrepancies were interpreted in terms of depths of weathering front, river incision and root network.

(14) Carefully selected small, yet watertight, catchment areas which are monitored for a number of years to account for climatic variability and supplemented by lysimetric plots if spatial variations in soils and vegetation require so, could give the best estimates of ecosystem (non-gaseous) macro-nutrient losses. The data base for tropical forest nutrient budgets must be considered weak, especially for nitrogen, and there is a need for more and careful studies in which specialists of various disciplines cooperate. Finally, standardisation of methodology and analysis should receive more attention in order to improve comparability of results.

(15) It is suggested that more effort be directed to the development of models describing hydrological and biogeochemical processes in those forests for which a good data set is available, rather than to start each time at new locations; such modelling is considered essential in improving our ability at predicting environmental consequences of forest conversions (cf. Beven 1988; Shuttleworth 1988b; O'Loughlin 1990; Vertessy et al. 1990).

(16) Apart from forests at specific locations, such as coastal fog belts or cloud belts in mountainous areas, or forests of very large areal extent (e.g. the Amazon basin), tropical forests most probably do not influence local amounts of rainfall significantly. Typical additions of moisture through cloud stripping amount to about 10 per cent of ordinary rainfall during rainy seasons but may well exceed amounts of rainfall during the dry season.

The best computer simulations of a complete conversion of the Amazon rain forest to degraded pasture have suggested a rise in temperature of about 2.5 °C and basin wide reductions in ET and precipitation of about 30 and 26 per cent respectively. Additional work is needed (and to some extent planned) to improve the land surface parametrisations of various land use types replacing the original forest, and their incorporation in global circulation models. In addition, little is known about the possible negative impacts on water yield of removing "cloud forests".

(17) With respect to the influence of forests on water yield (total streamflow) it is beyond doubt that both natural and (mature) man-made forests use more water than most agricultural crops or grass land (Table 4). Reported first-year increases in water yield following forest clearance range between 110 and 825 mm, depending on local rainfall.

(18) Converting lowland rain forest land to *well managed* grassland or annual cropping may produce permanent increases in total water yield of 200 - 300 mm yr^{-1}. On the other hand, one has to expect a more or less serious reduction in yield (depending on climate and soil depth) upon foresting degraded lands, particularly during dry seasons, unless the increase in ET associated with the replacement of shallow rooted short vegetation by a tall one is offset by a larger increase in infiltration capacity of the soil after forestation (which has yet to be reported). Further work is desirable.

(19) Converting lowland rain forest to tree plantations and extractive tree crops may (rubber?, cocoa?) or may not (oil palm, pines, eucalypts) lead to a permanent increase in water yield upon maturing.

Comparisons of on-site estimates of ET for mature rubber and cocoa plantations with the overall mean ET derived for natural forest suggested that permanent increases of 100 - 400 mm yr^{-1} might be possible. Additional work is needed to test this assertion. In the case of a conversion of natural forest to pine or eucalypt plantations, the temporarily increased water yields return to pre-clearing levels after canopy closure. Lysimetric work on water consumption of oil palm suggests a similar situation for non-irrigated trees but a substantial increase in ET (up to 400 mm yr^{-1}) upon regular irrigation. By contrast, fears that eucalypts are voracious water consumers is not supported by recent research, as long as the trees do not have direct access to the groundwater table.

(20) When evaluating the effects of the presence or absence of a forest cover on the magnitude of floods, the geological and climatic setting (together determining initial hillslope hydrological response) need to be taken into account : certain tropical forest areas have been shown to produce substantial amounts of storm runoff in the form of SOF (cf. (2)) whilst volumes associated with subsurface flow types (SSF) are generally much smaller. A shift from either of these runoff mechanisms to HOF may (SSF) or may not (SOF) produce significantly increased storm flow volumes and peakflows after forest clearance. In the former case, *relative* increases are largest for small storms and gradually smaller with increasing amount of rainfall.

(21) Although storm flow volumes roughly add up in a downstream direction, effects of locally increased flows are moderated by differences in time lag between tributaries and by spatial and temporal variations in rainfall. Truly widespread flooding is usually the result of an equally large field of extreme rain, occurring at a time when soils have become wetted up by previous rains. In such cases, the process of runoff generation is governed by soil water storage capacity rather than topsoil infiltration opportunities. The presence or absence of a well-developed vegetation cover has become of minor importance by then.

(22) Extreme flood events may also be related to the temporary blocking of a river by a large landslide and subsequent bursting of the dam (cf. Plate 12). Locally important factors include "backwater" effects near confluences, torrential rain on the plains themselves which becomes "trapped" on the fields when the river is in spate already, river training works (preventing lateral spreading), etc. Due to increased flood plain occupancy and economic growth, economic losses associated with major flood events have increased strongly in the last few decades. However, this should not be interpreted as if the actual magnitude of such events have increased as well.

(23) The apparent conflict between the often observed reduction in dry season flows following forest conversion to grazing or annual cropping in the real world and the increased flows reported for a number of controlled catchment experiments can be resolved by taking into account the net effect of changes in infiltration opportunities and ET associated with the respective land use types. If infiltration opportunities after conversion decrease to the extent that the increase in stormflow volumes exceed the increase in baseflow associated with reduced ET (real world situation), then dry season flow will decrease and vice versa (experimental basins) (cf. Plates 9 and 10). It should be realised, however, that any benefits of increased dry season water yield following clearing are often more than offset by increased stream sedimentation rates (see below). The importance of appropriate land husbandry after clearing in this respect will be evident.

(24) Where a well-developed litter or understory layer is present, surface erosion generally becomes minimal, whilst in the case of forested hillslopes the number of *shallow* (less than 1 m) mass movements is reduced as a result of the greater slope stability imparted by a well-developed tree root network. Reduction of these two forms of erosion is likely to be helpful in reducing gully erosion as well but often mechanical measures (checkdams, diversions) are needed in addition to vegetative measures (cf. Plate 14). Depending on the severity of deforestation and the associated disruption of soil surface and root network, these beneficial effects may be lost

again upon clearing. However, the presence or absence of a mature tree cover has no influence on the occurrence of deep-seated mass movements which are entirely controlled by geological and climatic factors (cf. Plate 12).

(25) As indicated above, applying soil conservation measures like contour cropping, bunding, grass strips, terracing, mulching, etc. or tree planting may well reduce amounts of sediment generated by surface erosion and shallow land slips entering the drainage system. However, due to storage effects, it may take several decades for basins larger than several hundred km^2 before stream sediment loads in the downstream parts will become noticeably smaller. For the same reason, effects of locally reduced sediment yields tend to decrease rapidly in a downstream direction, particularly in areas of moderate relief. As such, the frequently voiced claim that upland rehabilitation will solve most downstream problems, does require some specification of the spatial and temporal scales involved. Further work is needed in this respect.

(26) No substantial increases in soil nutrient levels or leachate concentrations associated with small tree-fall gaps have been reported for tropical rain forest areas. Amounts of solutes lost via leaching appear to increase above a discrete threshold, located somewhere between 200 and 500 m^2.

(27) Significant rises in streamflow and dissolved nutrient concentrations (notably of nitrate and potassium) have been observed after selective logging of forest in Malaysia and Surinam. Losses can be minimised by taking proper care of road layout and extraction techniques, maintaining an adequate riparian buffer strip, etc. The increases in nutrient losses reported to date are temporary, relatively small and should not create future productivity problems. More work at a variety of geological substrates is desirable, also with respect to sediment production and sources.

(28) Studies of the effects of natural tropical forest fires are rare. Results suggest effects may remain visible for several years after

the event as enhanced streamflow and rates of nutrient exports. Experimental studies have shown different responses to fire depending on soil type and ecosystem nutrient status. More work is needed.

(29) Although considerable amounts of nutrients are lost during the cropping period of the shifting cultivation cycle, the natural successional vegetation generally takes over quickly after abandoning a field. Prolonged cropping or abusive clearing methods will retard succession and therefore the build-up of nutrient capital in the regrowth. Shortening the fallow period too much, a phenomenon which is occurring more and more as a result of increasing population pressures, destabilises the system and adaptations will be required in such cases.

(30) Converting rain forest to pasture is associated with a serious reduction in site nutrient capital and this type of land use is generally not sustainable without inputs of fertilisers. Forest regeneration is usually possible after abandonment unless the soil has been mistreated severely. In general, the "margins" for grassland management are much narrower than those for forest land. Even in a degraded state, the latter will still be able to perform a protective function. Grassland on the other hand must be managed very well (controlled intensity of grazing and burning) if it is to maintain streamflow regimes adequately.

(31) Whole-tree harvesting seems possible on certain fertile soils, but on some of the poorer tropical soil types even partial harvesting (e.g. stemwood plus bark only) will not be sustainable without additions of fertiliser. More work is needed, covering new combinations of tree species and soil types.

(32) Although permanent annual cropping on infertile rain forest soils has been demonstrated to be sustainable with carefully planned agronomic techniques, it is doubtful whether these can be applied widely in remote tropical areas. Agroforestry systems hold considerable promise in this respect but their water and nutrient dynamics are only beginning to be explored.

(33) There is a need for more integrated (interdisciplinary) studies on the hydro- and soil chemical effects of forest conversion which should combine rigorous experimental design (e.g. paired basins or plots) with detailed process-oriented work. Recent work has shown that sophisticated equipment may be used for extended periods in rain forest areas without serious technical problems, provided the material is of high quality and expertly handled.

(34) Additional site-specific (regional) guidelines for land clearing operations on different soil types and under different climatic regimes are desirable although one would be well advised to follow the (effective) guidelines developed for the extreme climatic and pedological conditions prevailing in the coastal zone of Queensland (Cameron & Henderson 1979; cf. Pearce & Hamilton 1986). In general, existing prescriptions and guidelines could be adhered to more strictly.

(35) The information presented in this report leads to to the observation that the adverse environmental conditions so often observed following "deforestion" in the humid tropics are not so much the result of "deforestation" per se but rather of poor land use practices after clearing of the forest. *This is precisely where our hope for the future lies.*

(36) The above statement should not be taken to imply that the removal of forest is a harmless activity: the irretrievable loss of species and habitats is something highly undesirable and needs to be checked as well as possible. As pointed out earlier (no. 30), margins for managing tropical forest land, even when fairly degraded, are still very much wider than those associated with pastures or annual cropping.

(37) By taking watersheds (river basins) as principal planning units, a natural framework is created in which man's activities and their environmental impacts can be readily evaluated (also economically) in terms of land productivity and hydrological quality ("integrated watershed management"; cf. Carpenter 1983; Bower & Hufschmidt 1984).

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