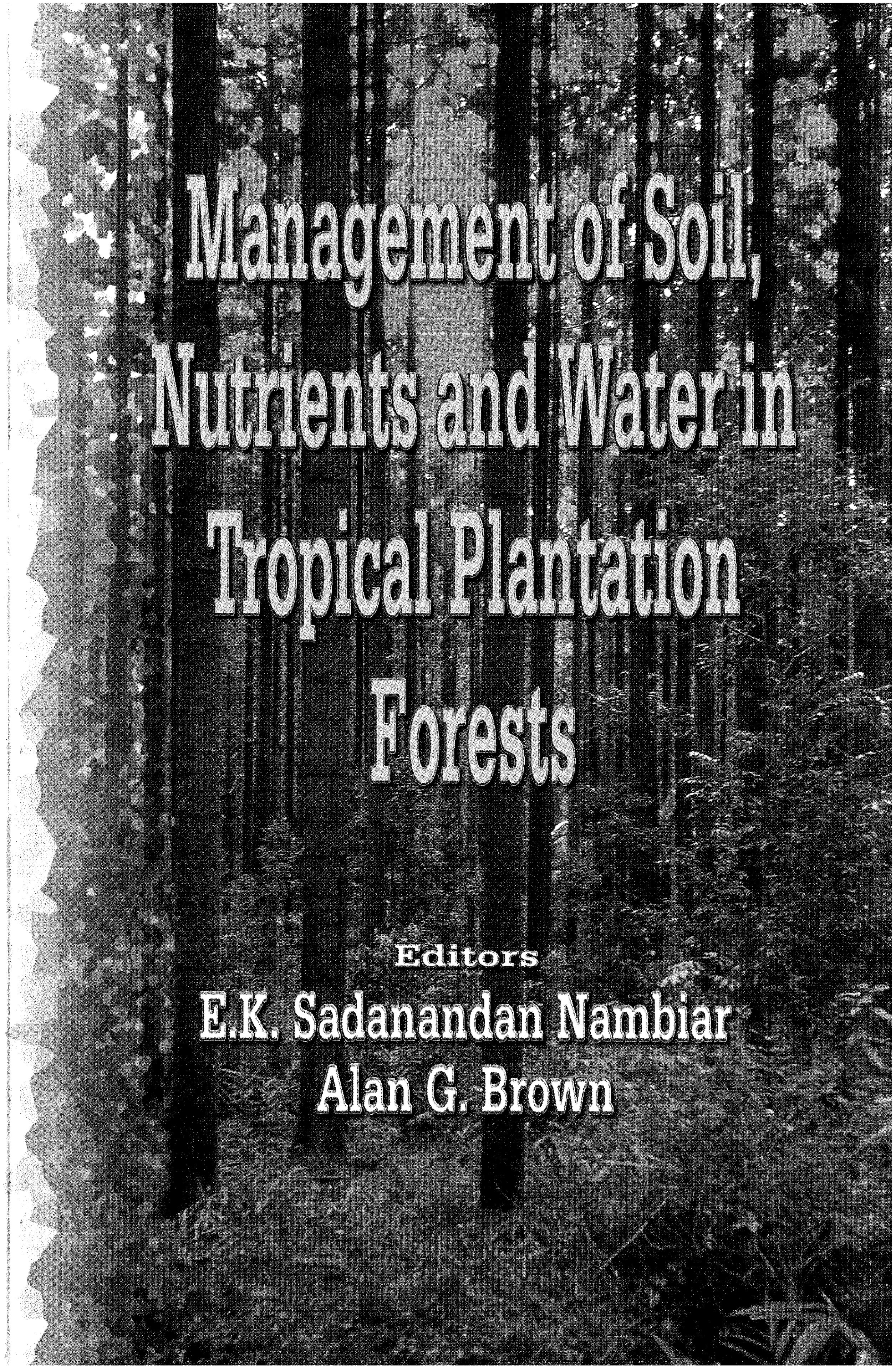
Management of Soil,
Nutrients and Water in
Tropical Plantation
Forests
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5

Hydrology of Forest Plantations in the Tropics

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Abstract

The stage of development of plantations, and their management, greatly influence their hydrological characteristics. The largest changes in catchment water yield, runoff response and sediment yield associated with plantations usually occur when natural forest is converted to plantation. Effects on peak streamflow rates and sediment loads normally stabilise within two yr of establishment at levels slightly above the original values. Comparatively little is known of the water use of even the most commonly-planted species. As a result, no sound predictions of the eventual effects of plantations on the magnitude of annual and seasonal water yield are possible at this stage. However, there is evidence that the planting of fast-growing trees on grassland will diminish streamflow after canopy closure, particularly during the dry season. The hydrological effects of forest management practices and the usefulness of catchment hydrological models in predicting such effects are discussed. It is concluded that, although relatively little is known at present of the hydrology of tropical plantations in relation to interactions between climate, soils and tree species, the methodology to fill the major gaps in knowledge is well established.

An UNDERSTANDING of the hydrology of plantations is important for several reasons—plantations are often situated in catchments which provide water for other land uses or urban purposes, and thus the quality and quantity of the water yield may be critical; the common combination of hilly topography and moderate or high rainfall at plantation sites is potentially conducive to erosion and/or operational difficulties, and not least the water relations of the plantation site have a profound influence on the choice of species (Chapter 2), nutrition and plantation growth.

Whilst in many parts of the humid tropics indiscriminate clearing of natural forest has led to serious degradation of soils and streamflow regimes, reforestation will not always lead to the restoration of the original hydrological conditions (Hamilton and King 1983). Improved flows have been claimed following reforestation (e.g. Hardjono 1980) but in other cases the planting of fast-growing trees on grass- and scrublands has considerably decreased dry-season water yield (Mathur and Sajwan 1978; Smith and Scott 1992; Waterloo 1994).

Concern has also been expressed about the potential loss of soil fertility by repeated harvesting of plantations (Hase and Fölster 1983; Russell 1983; Bruijnzeel and Wiersum 1985). The close linkage between the forest hydrological cycle and overall nutrient inputs and outputs is well-established and can be best evaluated by choosing a (watertight) catchment area as the fundamental unit (Bormann and Likens 1967). In addition, small catchments may also provide a suitable framework for the study of solute losses associated with forest disturbance and clearing (Likens et al. 1977).

This chapter aims to: i) provide a hydrological basis for the quantification of the nutrient budget at various stages of the plantation life cycle (as discussed further in Chapter 10); ii) reconcile conflicting evidence of the influence of plantations on water yield; and iii) examine hydrological impacts of various forest management options. The issues arising are examined and discussed from the viewpoint of general hydrological principles.

The account starts with a brief discussion of the forest hydrological cycle, followed by an analysis of the respective hydrological impacts (changes in water yield, infiltration capacity, peak flow and sediment yield) of different methods of forest clearing and site preparation. Next, the changes in hydrological characteristics (rainfall interception, stand water use) associated with the development and maturation of a plantation are discussed. This discussion is generally illustrated with data from the tropics, but where such information is lacking pertinent data from warm-temperate regions are used. Subsequently, the hydrological impacts of various forest management activities, and measures to minimise any adverse consequences during plantation establishment and harvesting, are considered. Finally, after identifying the most important gaps in our knowledge, a case is made for the establishment of a network of sites at which research can be concentrated on the most important species used in tropical plantations.

The Forest Hydrological Cycle

The principal features of the forest hydrological cycle are illustrated in Figure 5.1. Rain is the main precipitation input to the forests of the humid tropics. A small part of the rain reaches the forest floor as direct throughfall and stemflow. A substantial portion of that intercepted by the forest canopy is evaporated back into the atmosphere during and immediately after the storm; the remainder reaches the soil surface as crown drip. Because direct throughfall and crown drip cannot be determined separately in the field, the two are usually taken together and referred to as throughfall. If the intensity of the total throughfall and stemflow reaching the forest floor exceeds the infiltration capacity of the soil, the unabsorbed excess runs off as 'Hortonian overland flow' (HOF). Due to the generally very high absorbing capacity of the organic-rich topsoil in most native tropical forests, the bulk of the throughfall

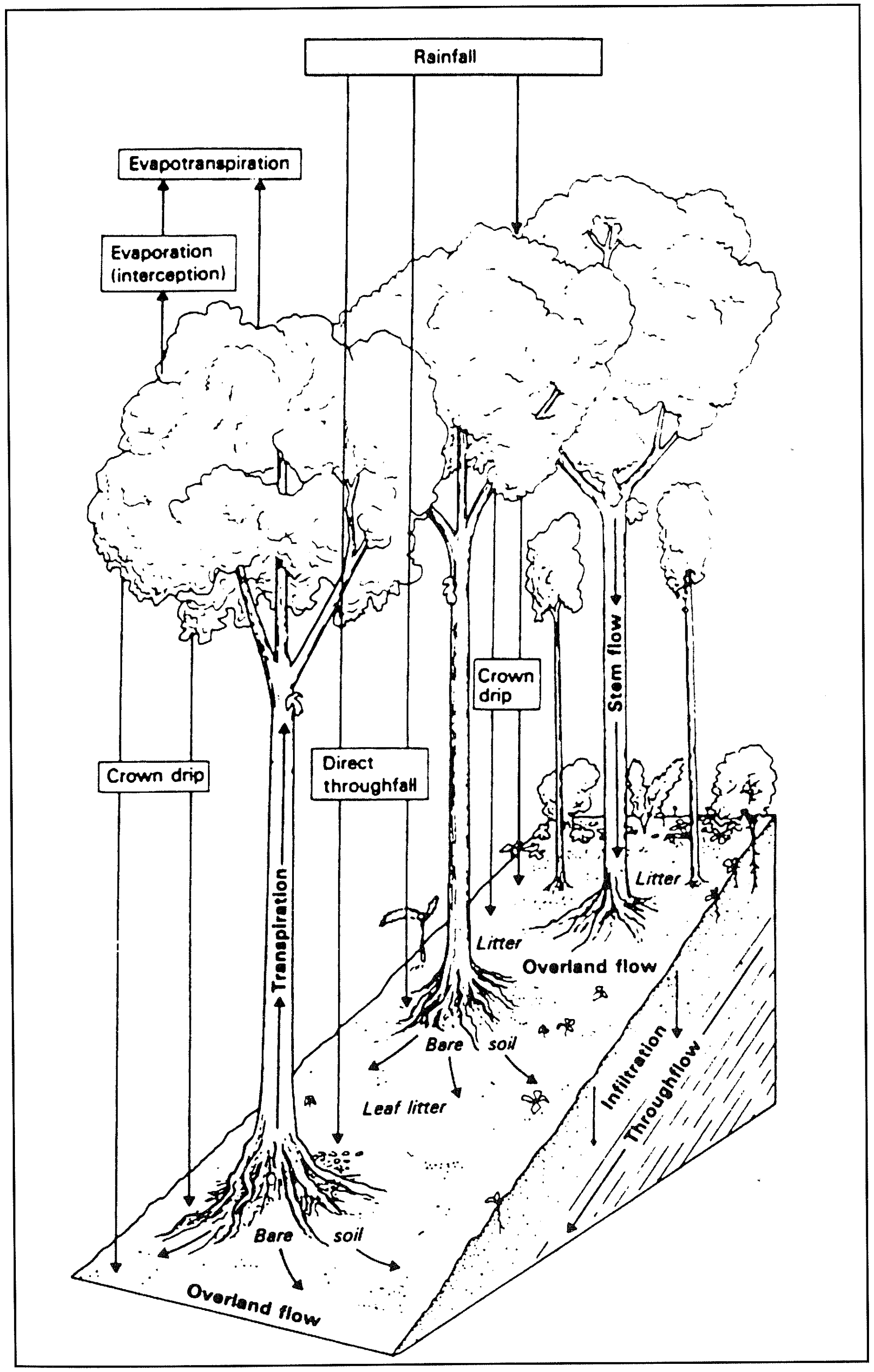

Figure 5.1 The hillslope forest hydrological cycle (modified from Douglas 1977)

and stemflow will generally infiltrate the soil and thus this type of flow occurs relatively infrequently. Much of the infiltrated water is taken up by the vegetation and returns to the atmosphere via the process of transpiration (Et). The term evapotranspiration (ET) is used to denote the sum of transpiration (evaporation from a dry canopy) and interception loss (Ei, evaporation from the exposed surfaces of a wet canopy). Evaporation from the litter and soil surface (Es) in (humid tropical) forest is usually negligible. It is important to make the distinction between Et and Ei because the former is governed largely by stomatal control and the latter mostly by the aerodynamic properties of the vegetation (Jarvis and Stewart 1979; Chapter 6).

The remaining soil moisture drains into the stream network by 'throughflow', the result of downward-moving water meeting an impermeable layer of subsoil or bedrock and being deflected laterally (Fig. 5.1). Such water drains slowly and steadily, thus accounting for the 'baseflow' of streams (Ward and Robinson 1990). In seasonal climates, baseflow reaches a minimum in the dry season and this is referred to as 'dry-season flow'.

During rainfall, infiltrated water may take one of several routes to the stream channel, depending on soil hydraulic conductivities, slope morphology and the spatial distribution of soil moisture (Dunne 1978). 'Saturation overland flow' (SOF) is caused by rain falling onto an already saturated soil. This typically occurs in hillside hollows or concave footslopes near the stream where subsurface flow tends to converge and so maintains near-saturated conditions. Additionally, SOF can be observed during and after intense rainfall where an impeding layer is found close to the surface (Bonell and Gilmour 1978). 'Subsurface stormflow' (SSSF) often represents a mixture of 'old' (i.e. already present before the start of the rain) and 'new' water travelling rapidly through 'macropores' and 'pipes' (Bonell and Balek 1993). As a result of contributions by SOF, SSSF, and in extreme cases HOF, streamflow usually increases rapidly during rainfall. This increase above baseflow levels is often called 'stormflow' or 'quickflow'. The highest discharge is commonly referred to as 'peak flow', and this may be reached during the rainfall event itself or as late as a few days afterwards, depending on catchment characteristics and wetness, as well as on the duration, intensity and quantity of the rainfall (Fig.5.2; Dunne 1978). The total volume of water produced as streamflow from a catchment over a given period of time (usually a month, season or year) is called 'water yield'.

The interlocked character of the chief components of the catchment hydrological cycle is illustrated by the water budget equation (Ward and Robinson 1990):

$$P = ET + Q + \Delta S + \Delta G \quad (1)$$

where
P = precipitation
ET = evapotranspiration
Q = streamflow
ΔS = change in soil water storage
ΔG = change in groundwater storage
with all values expressed in mm water per unit of time (day, week, month or year).

Furthermore:

$$ET = Ei + Et + Es \quad (2)$$

and

$$Ei = P - (Tf + Sf) \quad (3)$$

where
Tf = throughfall
Sf = stemflow

Detailed quantitative descriptions of the hydrological cycle of tropical rainforest have been given by Bruijnzeel (1990) and Bonell and Balek (1993).

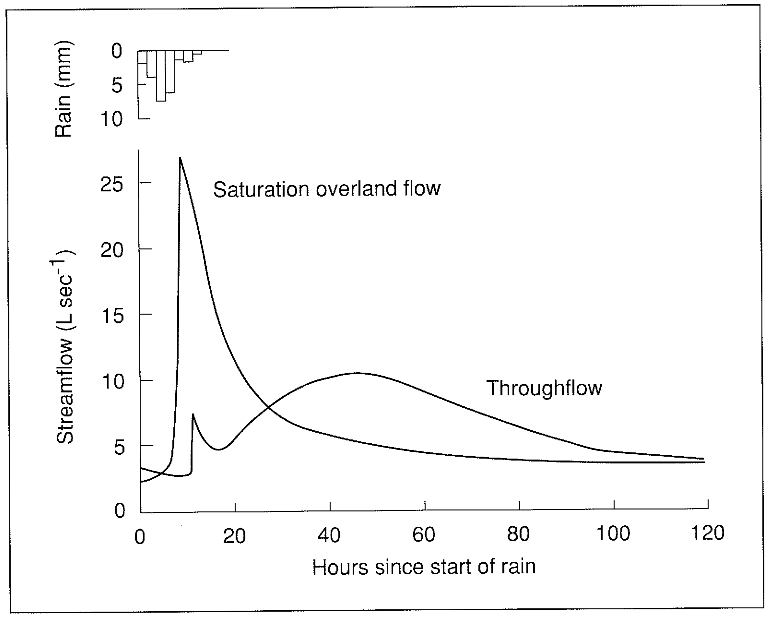

Figure 5.2 Catchment response to rainfall (modified from Calver et al. 1972)

Hydrology of Forest Plantations

The establishment of a plantation entails a number of practices which affect the hydrological properties of the soil, catchment hydrological response and tree growth. Generally, soil disturbance will be greater when plantations are established on natural forest land than when reforesting grassland. In the former case heavy mechanised equipment is often used to harvest large tree trunks and for site preparation, including the 'windrowing' of logging debris, whereas for reforesting grassland, burning and less intensive site preparation will suffice.

In view of the similarities between natural forest and mature plantations, it could be argued that the largest changes in hydrology associated with the conversion from one to the other can be expected during the first 1–3 yr after forest clearing and replanting, i.e. in the 'establishment' phase. The first 6–12 months of this are particularly critical because most of the soil surface will be exposed. Levels of peak discharge and erosion tend to return to predisturbance values within the next two yr due to the establishment of a vigorous understorey. However, the amount of water percolating through the soil (and hence amount of nutrients leached) will remain well above the original level during this period (cf. Malmer and Grip 1994). Subsequently, during 'stand development and maturation', there will be a gradual return to preconversion conditions. Hydrological changes during the establishment and maturation phases will therefore be discussed separately. When developing a plantation on grassland, the initial changes in hydrology will be smaller than those seen when starting with an already-forested site, while the maximum contrast will occur upon the stand reaching maturity (Smith and Scott 1992).

The establishment phase

Effect of clearing on soil physical properties

When heavy machinery enters undisturbed forest the protective litter layer will be disrupted. Topsoil bulk density will increase through compaction, particularly in the case of wet and clayey soils, and the water-holding and transmitting capacity of the soil will be reduced (see Chapter 4 for a general description).

Couper et al. (1981) compared several methods of forest clearing on an Alfisol in Nigeria in terms of man-hours and energy expenditure. Lal (1981) reported on the associated surface runoff and erosion rates for (no-tillage) agriculture during the first year after clearing. Whilst manual clearing was the slowest and most 'expensive', soil erosion was as low as 0.4 t ha^{-1} yr^{-1}, compared to c. 4 t ha^{-1} yr^{-1} after clearing with a crawler tractor with a shear blade, and 15 t ha^{-1} yr^{-1} after a crawler tractor with a tree pusher/root rake. Values of surface runoff were 1%, 6.5% and 12% of incident rainfall, respectively. More details are given in Tables 4.4 and 4.5 of Chapter 4.

Similarly, Dias and Nortcliff (1985b) reported negligible change in physical topsoil properties after traditional slash-and-burn clearance of a forest on a clayey Oxisol in Amazonia. However, considerable deterioration followed clearing by bulldozer, both because of compaction and because the removal of litter and topsoil exposed subsoil with less favourable properties (cf. Gillman et al. 1985; Malmer and Grip 1990). On clayey Oxisols and Ultisols, Dias and Nortcliff (1985a) and Kamaruzaman Jusoff (1991), respectively, reported a close relationship between the number of tractor passes and the resulting degree of compaction. In both cases, soil deterioration increased with soil wetness. Wheeled machinery had considerably greater impact than tracked vehicles (Fig. 5.3), particularly after wheel slippage. Effects usually extend to 15–20 cm depth and, in the case of former log haulage tracks, show very little improvement even after many years of nonusage (Hendrison 1990; Van der Plas and Bruijnzeel 1993).

Because of their skeletal structure, sandy soils show a different response to disturbance. Mechanised clearing of forest on podzols in East Malaysia did not result in a major increase in topsoil bulk density although final infiltration rates were reduced considerably (from 48.7 down to 1.3 mm hr^{-1}; Malmer and Grip 1990). However, with the exception of valley bottoms, saturation will rarely occur in sandy soils. It is important, therefore, to examine the effects of soil disturbance over a range of moisture conditions, as represented by soil water retention and unsaturated hydraulic conductivity curves. Jetten (1994) compared such curves for two types of sandy soils in Guyana before and after disturbance by wheeled skidders. White sand soils were most affected by repeated passage of machinery. Their retention curves became more gradual, i.e. suction values for a given moisture content increased as a result of a shift in pore distribution towards finer pores. Unsaturated hydraulic conductivities decreased accordingly, particularly at the wetter end of the suction range. The explanation may lie in the compaction of the originally loosely packed sand but possibly also in the mixing of fine organic matter with soil mineral particles. The effect was less pronounced for loamier brown sand soils, but here the changes occurred over the entire moisture range (Jetten 1994). Although manual methods of forest clearing cause less damage to the soil surface than does mechanical equipment, there will often be no choice but to resort to mechanical means.

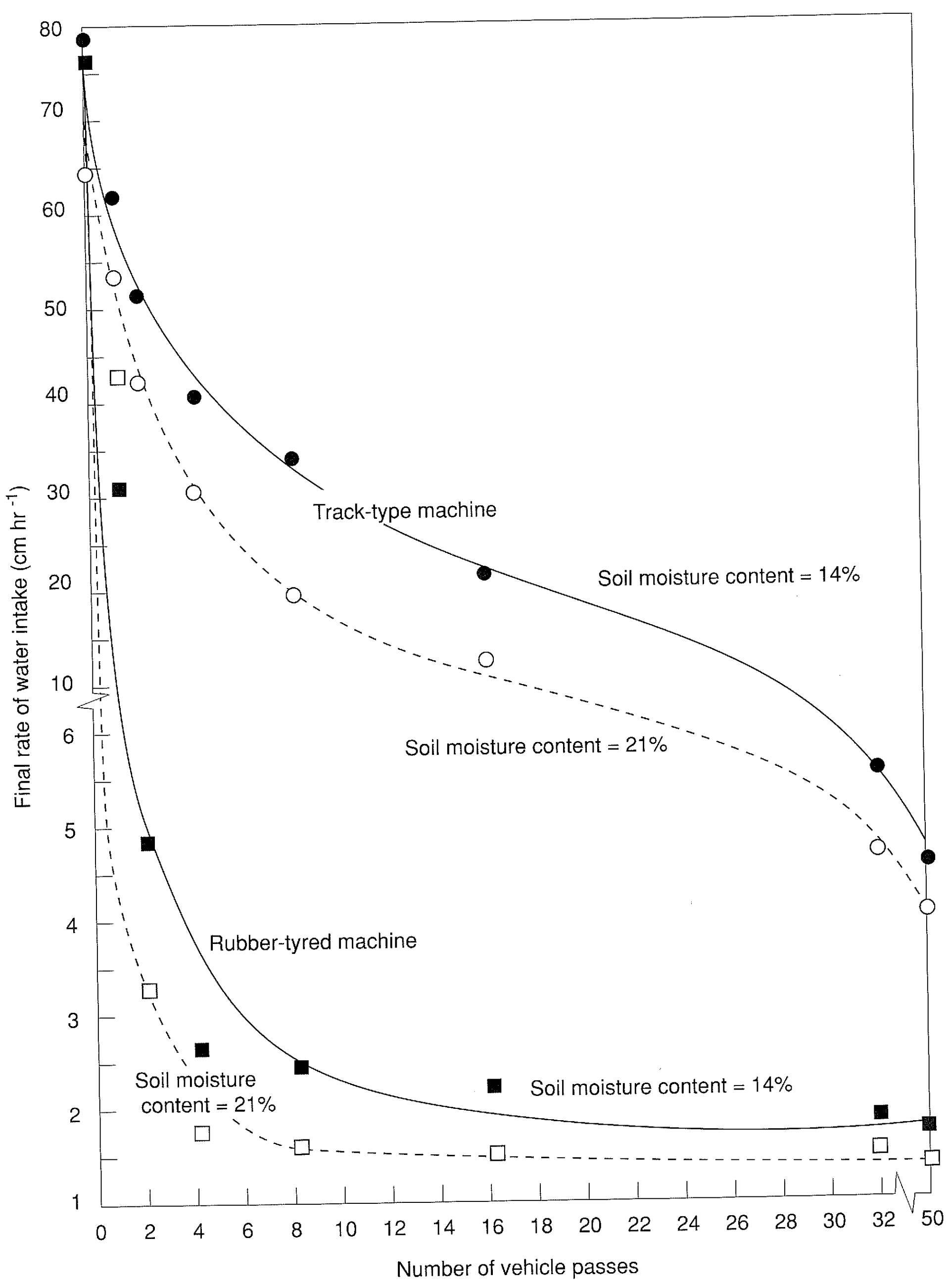

Figure 5.3 The impact of wheeled and tracked vehicles on saturated topsoil hydraulic conductivity as a function of the number of vehicle passes and soil wetness for an Ultisol in Peninsular Malaysia (adapted from Kamaruzaman Jusoff 1991)

Effect of clearing on catchment water yield

The removal of vegetative cover to establish a new plantation will temporarily reduce ET and hence increase catchment water yield. Bosch and Hewlett (1982) demonstrated this for about 100 cases from all over the world, including a few examples from the tropics. The most recent data set confirms that the initial increase in water yield is proportional to the amount of biomass removed (Bruijnzeel 1996a).

The reported increases in flow during the first three yr range from 125 to 820 mm yr^{-1}, with an average value of about 600 mm yr^{-1} (Bruijnzeel 1990, 1996). Whilst there is a weak tendency for larger increases to occur in high rainfall areas, the scatter in the data is large (Fig. 5.4). There are strong indications that the severity of soil disturbance is as important as post-clearing rainfall (Fig. 5.4). Malmer (1992) compared the changes in water yield in the first 2.9 yr of an *Acacia mangium* plantation after clearcutting rainforest in Sabah, East Malaysia, that had been selectively logged about six yr before the experiment. The different treatments were: 1) manual felling and extraction of timber followed by manual planting in rows between logging debris that had been left to rot; and 2) manual felling with timber extraction by crawler tractors, followed by burning of the slash (no windrowing) and planting. Whilst the increase in runoff was somewhat lower during drier years for both treatments (Fig. 5.4), overall increases were far smaller following manual extraction without burning than for mechanical harvesting plus burning (viz. 445 vs. 1190 mm; Malmer 1992).

Within the same experiment the secondary vegetation of an adjacent catchment that had been struck by an intense forest fire about five yr previously was burned again experimentally before planting it to *Acacia mangium*. The increase in streamflow over the same 2.9 yr was 1010 mm (Malmer 1992). Although these figures illustrate the potential for increased nutrient losses via leaching after clearing under wet tropical conditions, particularly where this is done in conjunction with burning (cf. Bruijnzeel 1996b; Chapter 10), it is impossible to separate the effects of mechanised clearing and burning in this particular experiment (see Bruijnzeel (1995) for details).

Effects of clearing on stormflow volume and peak discharge

The magnitude of the various flow components contributing to stormflow will vary between catchments as a result of differences in topography, soils (notably hydraulic conductivity) and rainfall characteristics (Dunne 1978; Pearce et al. 1982). The effect of the soil factor is illustrated by the results obtained for ten small (<1.5 ha) rainforested catchments in the perhumid (annual rainfall ca. 3500 mm with no clear dry season) lowlands of French Guyana, all close to each other (Fig. 5.5). Storm runoff ranged between 7.3% of rainfall (catchment h) and 34.4% (catchment e),

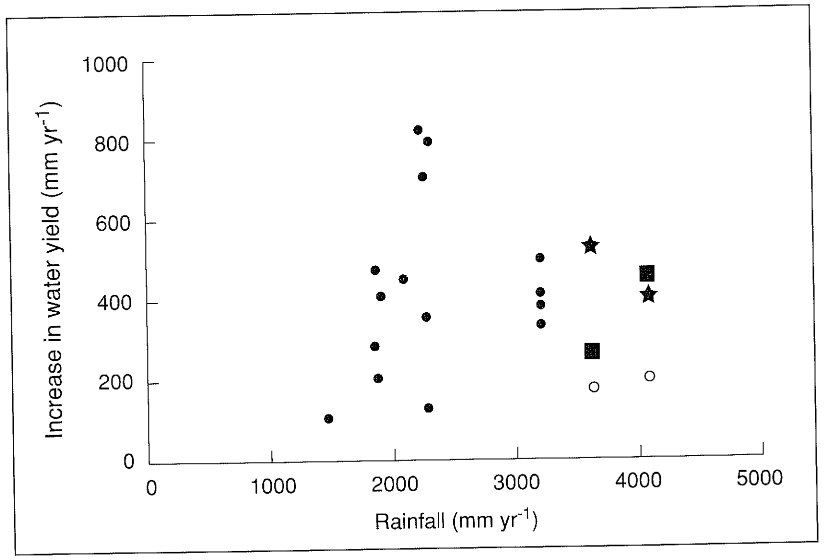

Figure 5.4 Increases in water yield after clearing tropical forest vs. corresponding amounts of rainfall (modified from Bruijnzeel 1995). The results for three adjacent catchments in East Malaysia (W1, W4 and W5), in the two yr following clearing in different ways are identified: Catchment W1, nonmechanised clearing followed by burning of slash = ★; catchment W4, manual extraction of logs, no burning = ○; catchment W5, mechanised harvesting, followed by burning of slash = ■ (Malmer (1992). Observations from other studies = ●

depending on soil type. The proximity of the groundwater table to the surface in the valley bottom at the height of the rainy season governed catchment response, and the larger the proportion of well-drained soils the smaller the runoff response (Fritsch 1992).

Normally, streamflow peaks that are produced by overland flow tend to be more pronounced than those generated by slower-travelling subsurface flow (Fig. 5.2). A shift from a slow throughflow-dominated stormflow pattern in forest to one that is governed by HOF after clearing may produce substantial increases in peak flows. However, in catchments where overland flow is already rampant under natural conditions, the hydrological response to rainfall (but not that of soil erosion) is hardly changed after forest removal (Gilmour 1977). Such a situation may occur where an impeding layer at shallow depth leads to widespread hillslope SOF (Bonell and Gilmour 1978) or pipeflow (Elsenbeer and Cassel 1990). All this points to the need for proper pedo-hydrological (in addition to standard soil) survey for assessing the hydrological impact of forest management practices.

Carefully planned and conducted conversion operations will be able to keep the area of compacted soil, and hence increases in the frequency and magnitude of HOF, to a minimum, particularly in the absence of burning (Hsia 1987; Malmer 1993). However, even with minimum soil disturbance, stormflow volumes and

peak flow rates may increase after forest removal. This is because the associated reduction in ET will lead to wetter soils which will be less capable of accommodating additional rainfall. In addition, this extra moisture will maintain a wider saturated strip along the stream which, in turn, will yield a larger volume of (SOF-dominated) storm runoff. Wherever topography permits expansion of this riparian 'source area' sufficiently, the effect will be noticeable, particularly after dry periods when under forested conditions the source area will have reached a minimum (Hewlett and Doss 1984; Fritsch 1992). Estimates of the increases in stormflow after clear-cutting with minimum soil disturbance may be derived from an experiment in a high rainfall area in New Zealand where runoff was dominated by subsurface flow (Pearce et al. 1980). Relative increases (as compared to a forested control basin) in stormflow volume were 100%–300% for small rainfall events (say <10 mm); c. 50% for intermediate storms (25–50 mm); and 10%–25% for large (>75 mm) storms. The effect is thus seen to diminish as rainfall increases. In a similar experiment in a steep catchment in Taiwan, no significant change in stormflow volume was observed but the median value of peak discharge rate increased by almost 50% as a result of the increased wetness of the hillslopes (Hsia 1987).

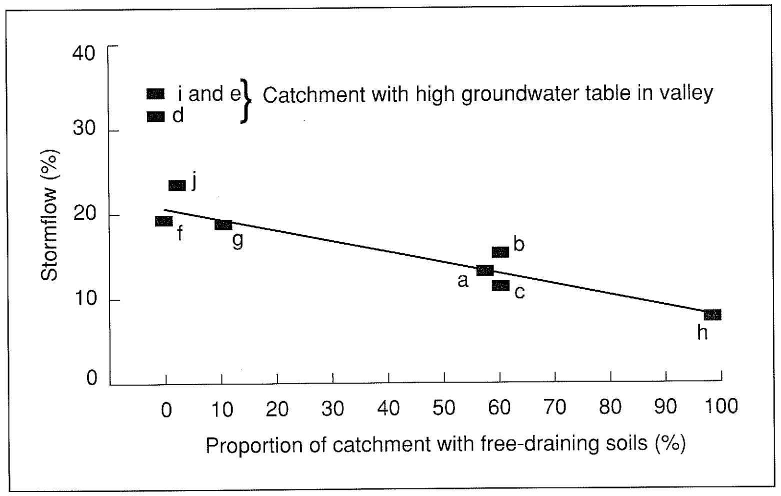

Figure 5.5 Runoff response (stormflow as a percentage of annual rainfall) of small forested catchments in French Guyana as a function of the proportion of catchment area underlain by free-draining soils (modified from Fritsch 1993)

When soils are more extensively disturbed by heavy machinery and/or fire for plantation establishment, the resulting changes in catchment response can be expected to be more pronounced. Yet, few unambiguous results have been obtained in this respect, possibly because the complexity of the runoff generation process often defies the simplicity of our analytical techniques (Hewlett 1982). For example, in the experiment by Malmer (1992) in East Malaysia, both stormflow volumes and peak discharges increased significantly during the first two yr after burning a catchment that had been struck by a natural forest fire several years previously. On the other hand, the only significant change that was observed after mechanically clearing the vegetation and burning the slash in a nearby catchment, was a reduction in stormflow volume during the first year, even though the soil had become severely disturbed over 24% of the area (Malmer and Grip 1990). During the second year after clearing, stormflow volumes and peak discharges increased again but not significantly. Malmer (1992) interpreted these findings in terms of contrasts in surface detention characteristics between the two catchments. For instance, water was initially seen to be standing in the ruts of former tractor tracks which then started to erode during the second year. Similarly, Swindel et al. (1983a) reported a sixfold increase in peak discharges immediately after mechanically clearing and burning a *Pinus elliottii* flatwood forest in north Florida. This was followed by a gradual decline over the next 18 months. Yet, no changes were observed in the correspondingly increased stormflow volumes throughout the two-and-a-half yr post-clearing observation period (Swindel et al. 1983b). Normally, such increases tend to diminish within a few years as the new vegetation becomes established, unless they are associated with structural changes in the runoff generation process, for example due to the construction of roads or the creation of active gullies. These findings point to the value of process studies in support of the traditional 'black box' paired watershed approach.

A major experimental investigation into the effects of tropical forest conversion on stormflow volumes, peak discharges and erosion has been conducted in French Guyana (Fritsch 1992, 1993). Ten small forested catchments with different admixtures of freely- and/or poorly-drained soils were gauged for two yr (cf. Fig. 5.5) before being subjected to a variety of treatments, including forest regeneration after logging or slash-and-burn agriculture, and conversions to grassland or plantations of *Pinus caribaea* and *Eucalyptus grandifolia*, whilst two catchments remained undisturbed as controls (Fritsch 1992, 1993).

Although the different hydrological behaviour of the respective catchments under undisturbed conditions (Fig. 5.5) precludes a direct comparison of the effect of each treatment, several interesting comparisons are possible (Fig. 5.6). For example, the similarity in predisturbance runoff response of catchments a, b and c (Fig. 5.5) enables an evaluation of the effects of logging (catchment a), mechanical clearing (catchment b) and traditional clearing (catchment c). Logging followed by regeneration produced only minor increases in stormflow (2%–4%), with a maximum increase (26%) during the (wet) third year and no detectable increase in the fifth year (Fig. 5.6a). Peak discharge increased (by 13%) during the first year only. In contrast, where logging was followed by mechanical clearing and burning, stormflow volume increased by about 150% during the first year (Fig. 5.6b). With

the establishment of the regrowth in subsequent years, this value dropped rapidly to 40% in the second year and to 16% in the fourth year. Peak discharge showed a marked increase only during the first year (42%) but not during subsequent years (Fritsch 1992). Changes in both stormflow volume and peak discharge associated with slash-and-burn were 26–30% during the two yr of cultivation (Fig. 5.6c). Although these values are much smaller than the ones observed immediately after mechanical clearing, there is nevertheless a clear effect.

The two catchments converted to forest plantations (d and e) showed the greatest runoff response in the undisturbed state (Fig. 5.5). The relative increases in stormflow volume or peak discharge after clearing (Figs 5.6d and 5.6e) may therefore look smaller than those in some other cases. The absolute increases during the first rainy season after clearing on these two catchments were however quite high (viz. 560–620 mm compared to 245 mm for catchment b where natural regrowth was allowed; Fritsch 1993). The increase in hydrological response decreased with time (Figs 5.6d and 5.6e), although not as fast as in the case of natural regrowth after clearing (Fig. 5.6b), as the plantations had to be weeded several times during these early years. The corresponding increases in peak discharges were 55–65% during the first two yr but declined rapidly afterwards (Fritsch 1992). By the sixth year, increases in stormflow had become negligible (Figs 5.6d and 5.6e).

It is of interest to also examine the effects of conversion from forest to grazing land on catchment response because this could indicate what may be expected after the reforestation of grassland. In the undisturbed state, the catchment that was to be converted to *Digitaria* grassland (catchment f) showed a runoff response of intermediate magnitude (Fig. 5.5). Following conversion, increases in stormflow volume remained substantial throughout the first four yr (generally >50%), with a somewhat lower value (27%) in the fifth year (Fig. 5.6f). Levels of peak discharges were consistently about 75% higher than expected under forested conditions (Fritsch 1992). These figures illustrate the considerable reductions in runoff that may follow reforestation of tropical grasslands.

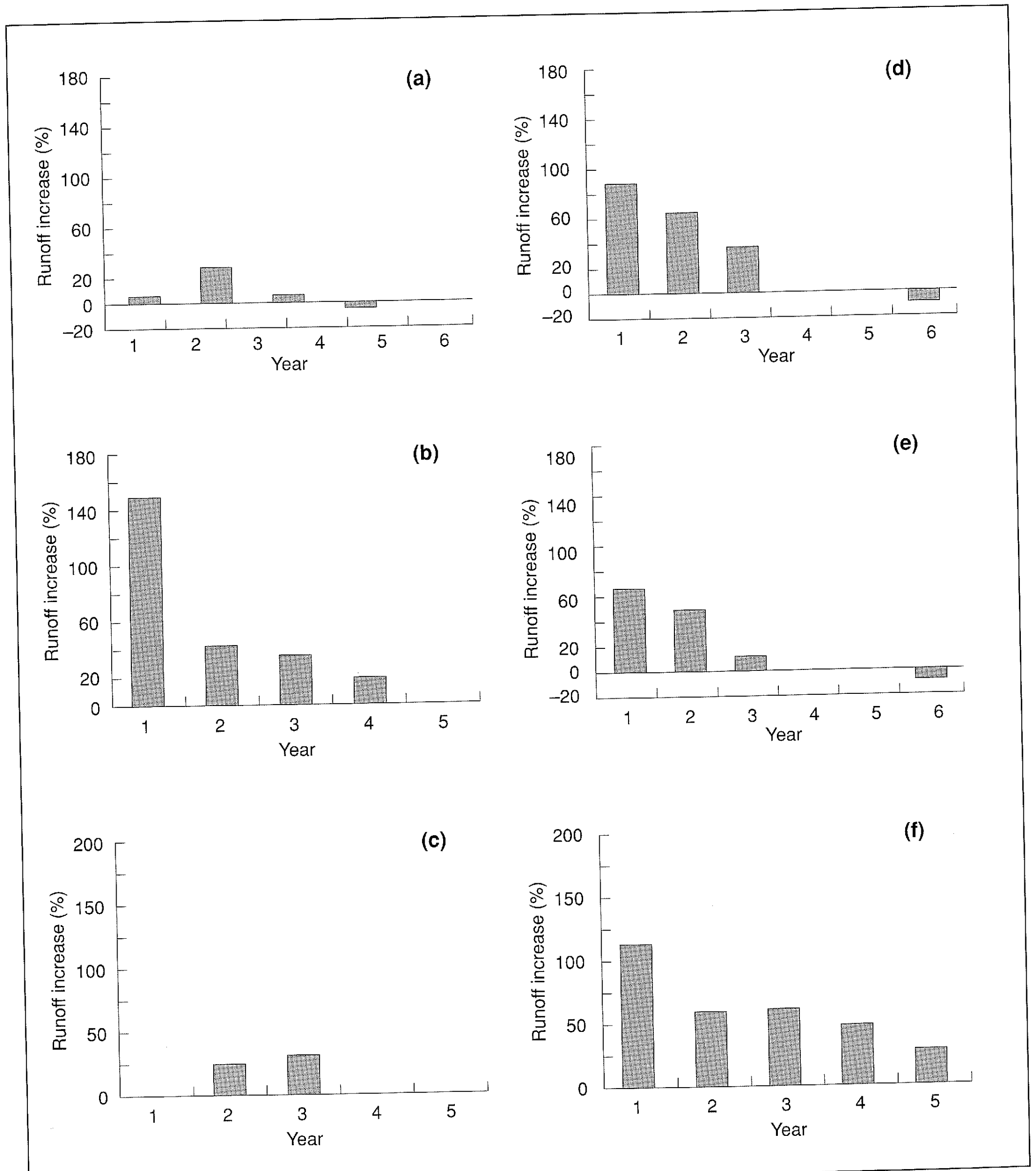

Figure 5.6 Increases in storm runoff after subjecting lowland rainforest in French Guyana to: a) logging followed by regeneration; b) mechanised clearing and burning, followed by regeneration; c) slash-and-burn cultivation; d) clearing and burning, followed by establishment of *Pinus caribaea*; e) idem for *Eucalyptus grandifolia*; f) idem for *Digitaria* grassland (modified from Fritsch 1993)

Effect of clearing on basin sediment yield

There are several different forms of erosion. Splash erosion is the process by which soil particles are detached by the impact of rain drops on the soil surface. The eroded particles, which may have been moved only a few centimetres, are then susceptible to further transport downslope by overland flow (sheet erosion). Both splash and sheet erosion are of little importance in most undisturbed forests, but they may well produce substantial amounts of sediment after the soil is bared. Once this stage is reached, topographic irregularities often lead to the concentration of overland flow in 'rills'. If the process continues long enough, these rills may deepen and widen into gullies. 'Mass wasting' is another mechanism of sediment supply to streams, especially common in steep areas where rainfall is high. Landslips and riverbank erosion fall into this category, and are often a natural hazard. The magnitude of sediment production under forested conditions depends on the relative importance of the respective contributing mechanisms (Pearce 1986). Sediment yield from rainforested catchments may be as low as 0.25 t ha^{-1} yr^{-1} in stable areas underlain by deep permeable soils that are neither subject to significant sheet erosion nor mass wasting (Douglas 1967). Yet, in tectonically active tropical steepland areas prone to hillslope failure, such as are frequently found along the Pacific rim, this figure may well approach 40 t ha^{-1} yr^{-1} during wet years (Dickinson et al. 1990) or more in specific cases (e.g. over 65 t ha^{-1} yr^{-1} for unstable marly soils; Van Dijk and Ehrencron 1949; Bell 1973). Any effects of forest clearing will be most evident where natural rates of sediment production are low.

It is important to recognise that not all eroded material is delivered directly into the stream network. Particles are often trapped temporarily (or permanently) in depressions in the terrain or deposited on footslopes or alluvial plains. This is especially true for splash and sheet erosion, and explains why it is not possible to predict catchment sediment yields from observations of erosion made on small hillside runoff plots alone. On the other hand, gully erosion, large landslides and riverbank erosion deliver sediment directly into the stream. The arrival of sediment at the stream network or its immediate surroundings, however, does not mean it will appear downstream immediately. An exceptional peak flow may be necessary to wash previously-deposited sediment into its final resting place. Thus the effects of soil disturbance tend to be first manifest on hillslopes in the form of increased sheet erosion and only later as increased catchment sediment yield (Walling 1983; Bons 1990).

Whilst it is difficult to quote 'typical' values for increases in hillslope erosion and catchment sediment yield accompanying forest conversion in the humid tropics, an idea can be gained from the results obtained by the studies by Malmer (1990) in East Malaysia and by Fritsch and Sarailh (1986) in French Guyana. Both experiments took place in high rainfall areas (>3500 mm yr^{-1}) with low background sediment yields before clearing.

The changes in yield of suspended sediment from the catchment associated with the conversion of rainforest to a plantation of *Acacia mangium* in East Malaysia can be summarised as follows. Pre-disturbance values were very low at 0.05 and 0.2 t ha^{-1} yr^{-1} for two catchments, W4 and W5, respectively. Catchment W4 was then cleared manually without burning of slash whilst catchment W5 was cleared mechanically and burned. Increases in sediment yield during the period of clearing plus the next five months necessary to reestablish the understorey in catchment W5 were six- and tenfold for catchments W4 and W5, respectively. Surprisingly, in view of the different treatments, sediment production over the next ten months was equal in both catchments. Compared to the preceding eight-month period, this represented a threefold increase in the case of catchment W4 but a 50% reduction for catchment W5. Malmer (1990) interpreted the unexpected finding for catchment W4 in terms of suppressed understorey vegetation due to the presence of large amounts of slash. Also, the slash had been arranged in rows running down the slope and this may have influenced the occurrence of surface runoff as well. However, the overall effects were relatively shortlived. Catchment sediment yields were virtually back to normal within 20 months after clearing. Total sediment exports from the catchments during the first three yr after clearing were 2.3 and 4.1 t ha^{-1} for catchments W4 and W5, respectively. By contrast, surface erosion on former tractor tracks in catchment W5 exceeded 500 t ha^{-1} during the same period (Malmer 1993), suggesting that considerable volumes of sediment went into storage.

Sediment outputs from the catchments in French Guyana studied by Fritsch and Sarrailh (1986) ranged between 0.05 and 0.75 t ha^{-1} yr^{-1} before disturbance, depending on stormflow regime and soil type (cf. Fig. 5.5). Despite the very similar preclearing runoff behaviour of the two catchments converted to plantations of pine and eucalypt (catchments d and e; Fig. 5.5), increases in sediment yield during the first year after clearing differed markedly (50- and 16-fold increases to 17 and 6 t ha^{-1} yr^{-1}, respectively). This unexpected contrast can be explained by the fact that the valley bottom of catchment e had become surrounded by walls of earth and slash acting as a filter for sediment coming from upslope (Fritsch and Sarrailh 1986). An independent survey estimated average hillslope erosion at about 1200 t ha^{-1} (Fritsch 1983). Such contrasts between on-site erosion and off-site sediment yield underscore the point made earlier about possible discrepancies between the two (cf. Bons 1990). No information has been published on sediment yields for these catchments during subsequent years but observations for a nearby catchment converted to grassland suggested a rapid decline in sediment yield during the second and third year after clearing (to a stable value of about 0.5 t ha^{-1} yr^{-1}, or three times the value expected under forested conditions), compared to an initial 34-fold increase to 12 t ha^{-1} yr^{-1} during the first year (Fritsch and Sarrailh 1986).

Much more modest increases in sediment yield (i.e. less than double the values observed under forest) may be obtained when using skyline log extraction techniques, as has been demonstrated by Pearce et al. (1980) and O'Loughlin et al. (1980) in steep terrain in a high rainfall zone in New Zealand. These findings once more confirmed the importance of keeping soil compaction to a minimum during conversion operations.

The maturation phase

The initial changes in catchment runoff response and sediment yield associated with the early stage of a plantation become less pronounced as the new vegetation establishes itself, and these attributes usually assume stable values within 2–3 yr. However, as a result of increasing leaf biomass, both rainfall interception and forest water use (transpiration) will continue to increase, particularly until canopy closure. Certain fast-growing species, under humid tropical lowland conditions, may reach canopy closure within 3–4 yr (Lim 1988; Lugo 1992). Whilst increases in tree water use may level off when the growth rate stabilises (see Chapters 6 and 7), rainfall interception, and therefore total ET, may continue to increase as the plantation matures (Helvey 1967). In the following we will first examine the magnitude of the two principal components of evaporation, and then discuss the implications for total and seasonal water yield. Finally, the hydrological effects of various forest management practices will be treated.

Rainfall interception

There are numerous publications on rainfall interception in tropical forest plantations. However, comparison of interception data for different species and locations is complicated, not only because of potential contrasts in climatic conditions, tree growth, stand age and management history (thinning etc.) but also because of differences in methodology. In general, the amount of throughfall estimated with a 'roving gauge' method tends to be larger than that obtained with a 'fixed gauge' approach. This presumably reflects the different efficiency of the two methods in representing 'drip points' where amounts of throughfall exceed amounts of incident rainfall (Lloyd and Marques 1988). Because stemflow usually constitutes only a minor fraction of the total amount of water reaching the forest floor, it follows that the higher the throughfall, the lower the resulting interception (cf. Equation 3).

Results from what may be the most reliable interception studies (e.g. because of the use of a roving gauge technique and/or a large number of collectors plus a sufficiently long (>1 yr) observation period) for several important species used in tropical forest plantations have been collated in Tables 5.1 (hardwoods) and 5.2 (softwoods). Several important contrasts between groups of species emerge from these data.

Table 5.1 Throughfall (TF), stemflow (SF) and interception (I) fractions of incident precipitation (P) in selected hardwood plantations and natural forests in the (sub) tropics

Species	Location	Age	Tree density (ha^{-1})	TF/P	SF/P	I/P	MAP[l] (mm yr^{-1})	Elevation (m)	Method of measuring throughfall
Fast growing hardwoods									
Eucalyptus tereticornis	India[a]	6	1660	0.81	0.08	0.12	1670	c. 700	4 fixed gauges, daily
Eucalyptus saligna	Brazil[b]	6	1685?	0.84	0.04	0.12	1280	540	12 roving gauges, daily
Acacia auriculiformis	Indonesia[c]		1010	0.81	0.08	0.11	3075	115	12 fixed troughs, daily
		5	1010	0.75	0.07	0.18			
Acacia mangium	Sabah, Malaysia[d]								
	Poor growth	4	1010[k]	0.86	?	≤ 0.14	3350	700	10 roving gauges, daily
	Reasonable growth	4	1360[k]	0.80	?	≤ 0.20			
	Good growth	4	1090[k]	0.72	?	≤ 0.28			
A. mangium	West Malaysia[e]	4 2/3	1110	0.62	0.04	0.35	2100	75	8 fixed troughs, weekly
		5 1/6	1705	0.57	0.04	0.39			Idem
Other hardwoods									
Swietenia macrophylla	Philippines[f]	15		0.79	0.01	0.20	2220	<1000?	4 roving gauges, daily
Tectona grandis	Philippines[f]	8		0.79	0.02	0.20	Idem	Idem	4 roving gauges, daily
T. grandis	Nigeria[g]	25		0.80	0.02	0.19	1200	210	5 fixed gauges, daily
T. grandis	India[h]	25		0.73	0.06	0.21	1670	700	8 roving troughs, daily
Natural forest									
Lowland rainforest	Malaysia[i]	–	–	0.81	0.02	0.17	2825	220	40 roving gauges, daily
Lowland rainforest	Brazil[j]	–	–	0.91	0.02	0.07	2475	100	36 roving gauges, weekly

[a]George (1978); [b]Lima (1976); [c]Bruijnzeel and Wiersum (1987); [d]A. Malmer, pers. comm.; [e]Lai and Salleh (1989); [f]Castillo (1984); [g]Okali (1980); [h]Dabral and Subba Rao (1968); [i]Sinun et al. (1992); [j]Lloyd and Marques (1988); [k]density at age 22 months, Sim and Nykvist (1991); and [l]mean annual precipitation

Table 5.2 Throughfall (TF), stemflow (SF) and interception (I) fractions of incident precipitation (P) in selected softwood plantations and natural forests in the (sub) tropics

Species	Location	Age	Tree density (ha^{-1})	TF/P	SF/P	I/P	MAP[o] (mm yr^{-1})	Elevation (m)	Method of measuring throughfall
***Pinus* spp. (lowlands < 750 m)**									
P. caribaea	Brazil[a]	6	?[l]	0.90	0.03	0.07	1280	540	12 roving gauges, daily
P. caribaea[b]		13	700	0.88	–	≤0.12	1300	<500	10 roving gauges, weekly
P. caribaea	Fiji[c]	6	825	0.79	0.01	0.20	1800	115	20 roving gauges, daily to weekly
		11	820	0.77	0.01	0.22		45	
		16	620	0.82[n]	0.01	0.17		80	
P. merkusii	Indonesia[d]	9	710	0.73	0.04	0.23	2100	80	10 roving gauges, daily for 4 months
***Pinus* spp. (uplands > 750 m)**									
P. merkusii	Indonesia[e]	31	560	0.72	<0.01	0.28	2120	1375	20 roving gauges, daily
P. kesiya	Philippines[f]	10–15	704	0.84	0.06	0.11	3525	c1500	4 roving gauges, daily
P. kesiya[g]		30	?[m]	0.85	0.02	0.13	3600	1365	3 roving troughs, daily
Other coniferous softwoods									
Araucaria cunninghamii	Northeast Australia[h]	42	664	0.75	–	≤ 0.25	1560	760	20 fixed gauges, weekly
		42	764	0.81	–	≤ 0.19	2100	700	
Cupressus macrocarpa	Kenya[i]	20–25	?	0.75	–	≤ 0.25	2235	2650	41 fixed gauges, daily
Other softwoods									
Albizia falcataria	Indonesia[j]	5–6	600	0.82	–	≤ 0.18	3075	100	10 fixed troughs, daily
A. falcataria	Philippines[k]	8	?	0.77	0.03	0.20	2220	<1000	4 roving gauges, daily
Natural forest									
Submontane rainforest,	Queensland[h]	–	848	0.78	–	≤ 0.22	2100	760	20 fixed gauges, weekly
Montane rainforest,	Kenya[i]	–	–	–	–	0.20	2235	2850	41 fixed gauges, daily

[a]Lima (1976); [b]Lima and Nicolielo (1983); [c]Waterloo (1994); [d]Ruslan (1983); [e]C.A. Bons, pers. comm.; [f]Florido and Saplaco (1981); [g]Veracion and Lopez (1976); [h]Brasell and Sinclair (1983); [i]Pereira (1952); [j]Team Vegetation and Erosion (1979); [k]Castillo (1984); [l]original planting density 2 × 3m; [m]tree basal area 26m^2 ha^{-1}; [n]partly modelled; [o]mean annual precipitation

Rainfall interception by *Eucalyptus* spp. is modest at about 12% of incident precipitation, partly because throughfall and stemflow fractions are both relatively high (Table 5.1). Similar results have been obtained for natural eucalypt forests in southeastern Australia (Dunin et al. 1988; Crockford and Richardson 1990) and for plantations of *Eucalyptus camaldulensis* in a sub-humid part of southern India (Hall et al. 1992). As such, the frequently heard claim that the planting of eucalypts leads to depletion of water reserves (e.g. Vandana Shiva and Bandyopadhyay 1983) cannot be due to high interception losses. Of potential concern are the high interception losses reported for young fast-growing plantations of *Acacia mangium* in Malaysia (Table 5.1). Some of these high values may reflect the frequent occurrence of low-intensity rain (Abdul Rahim 1983), or perhaps a suboptimal sampling design, as in the example from Peninsular Malaysia (Lai and Salleh 1989). However, the results obtained by Malmer (pers. comm.) for vigorously growing *Acacia* plantations in East Malaysia (up to 28%) under high rainfall conditions are also considerable, even more so when expressed as absolute totals (up to 1150 mm yr^{-1}; Table 5.1).

Values for rainfall interception by such other widely planted hardwood species as teak (*Tectona grandis*) and mahogany (*Swietenia macrophylla*) are consistently around 20% over a range of climatic conditions (Table 5.1). Although teak is often leafless during the dry season, this is apparently more than compensated for by high interception losses during the wet season when leaf area indices may be as high as 6 $m^2\ m^{-2}$ (Grace et al. 1982). No interception data were found for *Gmelina arborea* but in view of its similarity to teak in terms of leaf area (Grace et al. 1982) and growth habit one would expect an average value of about 20% for *Gmelina* as well. The smooth bark of *Gmelina* may generate more stemflow than is found in *Tectona,* values for which are usually very low (Table 5.1); further work is needed.

Results obtained for various pine species are diverse (Table 5.2). Interception losses from rapidly growing stands of *Pinus caribaea* and *P. merkusii* were consistently above 20% of the rainfall at several lowland tropical locations (Fiji, Indonesia) but distinctly lower under more sub-humid and cooler conditions (southeast Brazil). High values (typically 25% or more) have also been reported for various coniferous plantations growing at higher altitudes not affected by frequent low cloud or fog. The latter results probably reflect a combination of low rainfall intensity prevailing at higher elevations (Calder 1990) and the high leaf area index associated with mature stands (Helvey 1967). Conversely, the low interception fractions obtained for *P. kesiya* in the mountains of the northern Philippines (0.10–0.13) may be influenced by contributions of 'occult' precipitation (mist) stripped by the trees but not recorded by the rainfall collector in the open (cf. Mamanteo and Veracion 1985). Finally, in view of its light crown, the values reported for the very fast-growing species *Albizia falcataria* seem rather high at 18%–20% (Table 5.2).

With the exception of Waterloo (1994), none of the studies quoted in Tables 5.1 and 5.2 have used above-canopy weather stations for the computation of evaporation rates from the wetted canopy in support of their traditionally-derived values of interception (cf. Lloyd et al. 1988). There is a need for more throughfall/interception studies in tropical forest plantations that combine a rigorous design for the measurement of throughfall and stemflow (cf. Lloyd and Marques 1988) with above-canopy observations of climatic parameters. Only in this way can we expect to be able to satisfactorily separate the influence of species and stand characteristics and climatic factors on the amount of rainfall intercepted.

Transpiration

The changes in water uptake by individual trees and at stand level during different stages in the plantation life cycle, and the underlying physiological principles, are dealt with in Chapter 6. Here, the available information on annual water use by plantations (including understorey vegetation) in the humid tropics, as determined by catchment hydrological and/or micrometeorological methods, will be examined. Because most studies have produced only short-term estimates of transpiration (e.g. Whitehead et al. 1981; Grace et al. 1982; Calder et al. 1992; Roberts and Rosier 1993), the data base for annual totals is limited (Table 5.3).

Whilst there is little point in comparing instantaneous transpiration rates for different species and age classes, especially as these may have been measured under contrasting climatic and soil water conditions, a few generalisations are possible. For instance, in most young to semimature stands in lowland areas, reported daily transpiration totals are in the order of 3-5 mm whenever soil water is not limiting, regardless of species (pine, teak, young eucalypts; Okali 1980; Roberts and Rosier 1993; Waterloo 1994). Occasional peaks to 6–8 mm day^{-1} may occur under particularly favourable soil water and radiation conditions; Kallarackal 1992; Waterloo 1994). In the case of older plantations, seasonally senescing leaves or progressive moisture stress, values usually drop to 1–3 mm day^{-1} (Monteny et al. 1985; Waterloo 1994), or less under very seasonal conditions (Roberts and Rosier 1993).

To facilitate comparisons between locations, values for Et and ET in Table 5.3 have been normalised by dividing them by the corresponding open water evaporation (Penman's Eo). Few general conclusions can be drawn from Table 5.3, however, except that the water use of mature coniferous plantations often resembles that of natural forest in the same area. More work is needed, particularly in broad-leaved forests.

The widespread idea that eucalypts are voracious consumers of water was critically examined by Calder (1992). He concluded that there were large variations in stomatal behaviour and rooting patterns, and thus transpiration rates, between different species of eucalypt. Whilst certain (naturally growing) species indeed exhibited little or no stomatal control, most eucalypts did. Eucalypts are therefore likely to transpire as much as other tree species, except in situations where their roots have direct access to the groundwater table (e.g. in depressions or valley bottoms). Under such conditions, very high evaporation rates may be maintained, particularly where atmospheric demand is high (Calder 1992). Young eucalypt plantations growing in a sub-humid part of southern India showed a distinct response to both increased air humidity deficit and soil water stress (Roberts and Rosier 1993). On the other hand, transpiration rates of coppiced 5-year-old *Eucalyptus tereticornis* trees growing in a wetter part of southern India (rainfall 2000 mm yr^{-1}) increased linearly with increases in air humidity deficit during the postmonsoon season when soil water was in good supply (Kallarackal 1992). Such observations point to the need for continued studies of the water use of *Eucalyptus* under conditions of high rainfall and deep soils.

Table 5.3 Annual evapotranspiration (ET) and transpiration (Et) of mature forest plantations and selected natural forests in the humid tropics. Values have been rounded off to the nearest five units

Species	Location	Elevation (m)	ET (mm)	ET/Eo	Et (mm)	Et/Eo
Agathis dammara[a,f]	Java	600	1070	0.79	405	0.30[i]
Pinus merkusii[b,f]	Java	1300	900	0.84	445	0.42
Pinus patula[c,f]	Kenya	2400	1160	0.77	600[h]	0.41[h]
Pinus caribaea[d] 6-yr-old[g]	Fiji	80	1770[j]	1.05	1250	0.74
P. caribaea[d] 15-yr-old[f]	Fiji	230	1510	0.90	1175	0.73
Lowland rainforest[e,g]	Java	<100	1480	0.90	885	0.54
Montane rainforest[c,f]	Kenya	2400	1155	0.78	690	0.46[h]

[a]Bruijnzeel (1988); [b]C.A. Bons, personal communication; [c] Blackie (1979); [d] Waterloo (1994); [e] Calder et al. (1986); [f] catchment water balance study, values for Et derived by subtracting interception loss from total ET and hence approximate only; [g] micrometeorological methods and/or Penman-Monteith evaporation model; [h] using interception estimate by Pereira (1952);[i] low value due to high interception loss (annual rainfall 4770 mm); [j] including c. 160 mm evaporated from litter layer.

Another example of consistently high water use concerns a 6-year-old plantation of *Pinus caribaea* in Fiji (Waterloo 1994; Table 5.3). Annual transpiration by this vigorously-growing stand was about 1250 mm. Adding amounts of rainfall intercepted by the canopy (360 mm) and the thick litter layer (160 mm) brought the annual total of ET to the very high value of 1770 mm. The corresponding amount obtained for a nearby 15-year-old stand was about 1510 mm. These values constituted 82%–92% of the rainfall during the study period and one can appreciate the concern of regional water authorities over rapidly dwindling streamflow following reforestation of grassland with pines in this particular area (Waterloo 1994). This and other examples will be discussed further below.

Plantations and total water yield

As a result of the increase in ET with stand age, streamflow can be expected to decrease with time after planting. Long-term streamflow records have been collected at several locations from which changes in water yield as the stand matures can be evaluated. For example, Blackie (1979) compared the water balance of montane rainforest with that for an adjacent plantation of *Pinus patula* on deep volcanic soils at 2400 m elevation in Kenya (cf. Table 5.3). During the first three yr after planting, total annual streamflow for the pine-forested catchment remained on average about 125 mm above that for the control catchment. Results for the subsequent six yr were somewhat variable (see Blackie (1979) for explanations) but once the pines were 10–11 yr old, streamflow patterns for the two vegetation covers started to coincide. It may be concluded, therefore, that the water yield from the pine-forested catchment remained above the original value over a period of 4–10 yr. Re-analysis of the data according to the more stringent paired catchment method (Hewlett and Fortson 1983) might reveal whether this figure would be closer to the 4-year mark or to the 10-year one.

Similarly, there is evidence that ET of mature stands of *Pinus merkusii* and *Agathis dammara* in the wet volcanic uplands of Java is similar to that of natural forest (ET/Eo = 0.80– 0.84; cf. Table 5.3). It remains to be seen, however, whether this will also be the case when old-growth lowland forest is converted to fast-growing plantations. Rainfall interception (e.g. *Acacia mangium*; Table 5.1) or total water use (e.g. young *P. caribaea* in Fiji; Table 5.3) may well exceed those of natural forest (typically 10%–20% of rainfall intercepted and ET between 1300 and 1500 mm yr^{-1}; Bruijnzeel 1990).

The issue of plantation forestry vs. water yield becomes more critical in areas where rainfall distribution is seasonal, particularly so if, in addition, soils are shallow and have little opportunity to store water (Bruijnzeel 1989; Smith and Scott 1992). Under such conditions the natural vegetation consists of deciduous forest, sclerophyllous scrub or grassland, with trees usually limited to the riparian zone. Samraj et al. (1988) and Sharda et al. (1988) reported on the effect on water yield of a partial (59%) conversion of a montane grassland catchment with swampy valley bottoms (left undisturbed) to a plantation of *Eucalyptus globulus* on deep permeable soils under seasonal rainfall conditions in southern India. During the first three yr, annual water yield was hardly affected but from then onwards stabilised around a level of 120 mm yr^{-1} (or 21%) below initial values until the trees were 10 yr old.

A particularly comprehensive data set has been collected under seasonal warm-temperate conditions in the Republic of South Africa. Here, a series of paired catchment experiments has been set up since the 1930s to evaluate the hydrological effects of afforesting grass- and scrublands in different parts of the country (Wicht 1967; Bosch 1982; Van Wijk 1987; Smith and Scott 1992). Figure 5.7

summarises some reductions in annual water yield after afforestation with *Pinus patula, Pinus radiata* and *Eucalyptus grandis*. Several points emerge from this comparison: i) the decreases in water yield follow a sigmoidal trend; ii) the effect of afforestation with *Eucalyptus* (Mokobulaan catchment) manifests itself more rapidly than that for pines; iii) the proportion of the catchment that is afforested influences both the rate (i.e. the slope of the curve) and the amount of change (e.g. Biesevlei vs. Bosboukloof which were situated in the same area). To some extent, the contrasts shown in Figure 5.7 are due to differences in site conditions. For instance, the relatively slow response observed for the Cathedral Peak II catchment was interpreted in terms of slow forest growth under harsh conditions (Smith and Scott 1992). Also, the reductions in flow after afforestation have been expressed in comparison to the flows associated with the original vegetation, which may have differed between locations. In some areas (e.g. Mokobulaan), this was 'dry' grassland with a low water use; in others (e.g. Biesevlei and Bosboukloof) it was socalled fynbos, a tall scrub vegetation with a higher water use than grassland (Smith and Scott 1992). Nevertheless, the salient contrast between eucalypts on the one hand and pines on the other was confirmed by two other paired catchment experiments in East Transvaal (not shown in Figure 5.7; Smith and Scott 1992). Typically, the planting of eucalypts produced a response in streamflow after about three yr (cf. the findings of Samraj et al. (1988) in India), whereas afforestation with pines only did so after about five yr. Because the water use of the eucalypts tended to stabilise after eight yr (cf. Fig. 5.7), the difference between the two genera became gradually smaller with time. For example, at the relatively dry location of Mokobulaan, Smith and Scott (1992) reported that the reduction in streamflow observed 11 yr after planting *P. patula* had reached a similar level to that observed for *E. grandis* three yr previously. Because effects of afforestation on stormflows were relatively small (Bosch 1982), the changes in overall water yield were primarily manifest during the dry season (see below).

Maximum falls in annual water yield after afforestation of grasslands under South African conditions were in the order of 400–500 mm (Fig. 5.7). Recent observations in Fiji indicate that the contrast in water use between plantations of *Pinus caribaea* and *Pennisetum polystachyon* grassland approaches such high values as 700–900 mm yr^{-1} (M.J. Waterloo and L.A. Bruijnzeel, unpublished data).

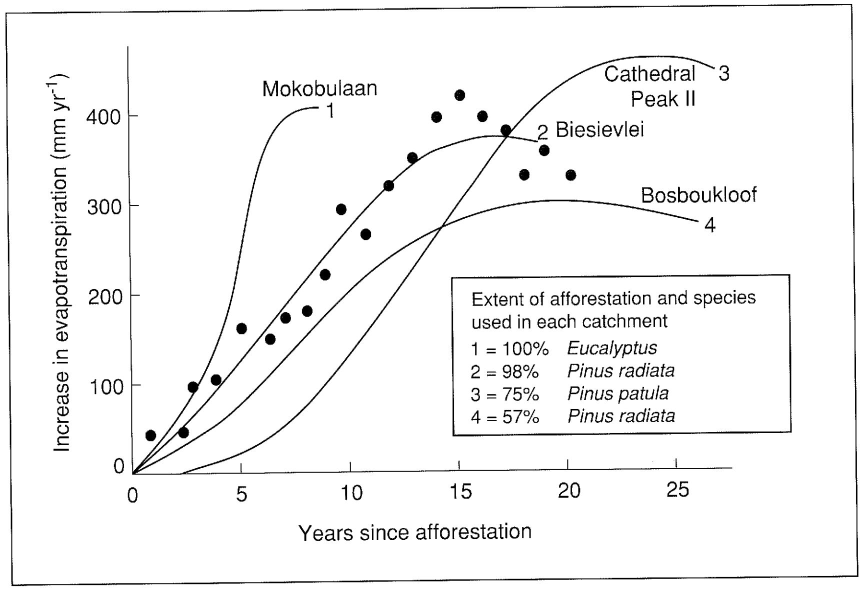

Figure 5.7 Increases in evapotranspiration with time after afforestation of grass- and scrubland in South Africa (modified from Bosch 1982). Actual data points are shown for the Biesievlei catchment only

Plantations and dry season flows

Contrasts in total evaporation between forest and grassland during wet periods mainly reflect the difference in rainfall interception between the two vegetation types (Calder 1990). During the dry season, however, the contrast in rooting depth between the two also comes into play. As a forest becomes older, the tree roots progressively reach greater depths whereas grasslands often die off during extended dry seasons and renew themselves during the next rainy season. As a result, the discrepancy in forest and grassland water use during the dry season tends to increase with age of the forest (Nepstad et al. 1994; Waterloo 1994; cf. Fig. 5.7). Therefore, depending on the vigour of the plantation, substantial reductions in dry-season water yield can be expected after afforesting grass- or scrubland. Whilst only a single study has documented the effect under humid tropical conditions (Waterloo 1994), the evidence strongly supports that obtained for warm-temperate to subtropical climates with a seasonal rainfall distribution (e.g. Smith and Scott 1992). Waterloo (1994) reported increases in dry season ET of 250–390 mm following the reforestation of fire-climax grassland in the dry zone of Viti Levu, Fiji, with *Pinus caribaea*, the actual amount depending on the age and vigour of the stand. These values correspond to a reduction in dry-season streamflow of about 50% (Waterloo 1994). In South Africa, Smith and Scott (1992) concluded that effects were noted earlier (usually after three yr) in the case of *Eucalyptus grandis* plantations than after planting *Pinus patula* or *P. radiata* (usually after about five yr). In addition, the effect turned out to be more marked for eucalypts (reductions of 90%–100%, regardless of average rainfall total or soil depth) than for pines (reductions of 40%–60%) during the first eight yr or so after treatment. However, as indicated already for total water yield, the difference—due to the different growth curves of the two groups of species—may disappear eventually, and thus the ultimate effect of afforestation on low flows may be the same, irrespective of species (Smith and Scott 1992).

At first, the experimental evidence presented above seems to contradict the deterioration in streamflow regime (increased flooding during rainy periods, reduced flow during dry spells) that is so often observed after deforestation in the tropics (Pereira 1989). The conflict can be resolved, however, by taking into account the net effect of changes in ET and rainfall infiltration brought about by the change in land use (Bruijnzeel 1989). Taking as an example a change from natural forest to degraded grassland, it will be clear that the total amount of streamflow will be much higher after deforestation (say, by 250–400 mm yr^{-1}) due to the decrease in water use associated with the transformation in cover. However, the timing of this increase in streamflow is strongly governed by the infiltration capacity of the soil. The potential surplus of water after forest removal can only manifest itself as increased baseflow during the next dry season if it has had the chance to infiltrate into the soil first. If, on the other hand, the capacity of the soil to absorb rainfall after deforestation is reduced too much by compaction or crusting, much of the water will run off down the slopes and contribute immediately to enhanced peak flows in the stream rather than percolate slowly through the soil and emerge later as baseflow. Under such conditions, dry season flows may well be reduced, despite the smaller uptake of water by the nonforest vegetation. If however infiltration capacity is maintained, the lesser water use of the new vegetation will show up as increased dry-season flow (Bruijnzeel 1989).

In view of the considerable difference in water use by seasonal grassland and actively growing forest (>250 mm yr^{-1}), the chance for increased dry-season flow after reforesting such lands is small, despite claims to the contrary (e.g. Hardjono 1980). Much depends on the ultimate rainfall-absorbing capacity following reforestation. One of the few studies that have looked at this important aspect is that by Gilmour et al. (1987) in the Middle Hills of Nepal. Gilmour and co-workers determined the saturated hydraulic conductivity of the soil under a range of conditions, viz. i) heavily grazed and trampled grassland; ii) a 5-year-old stand of *Pinus patula* on former degraded grassland; iii) and iv) 12-year-old stands of *Pinus roxburghii* on former grassland and scrubland, respectively; and v) protected natural forest. Average values of topsoil permeability increased from about 40 mm hr^{-1} (a low value) at the degraded grassland site to over 500 mm hr^{-1} in the protected forest.

When compared with the intensity of rainfall in the area, these data indicated that overland flow would actually occur on even the most degraded of sites as few as seven times a year, cf. a frequency of four times a year in the 5-year-old forest and zero occurrence in the older forests. Although topsoil infiltration capacities had improved by more than 140 mm hr^{-1}, 12 yr after reforestation, the associated reductions in surface runoff were small because of the relatively low rainfall intensities prevailing in the area. Moreover, they were much smaller than the 130–250 mm yr^{-1} of water that would be needed in this particular area to increase dry season flow (Bruijnzeel and Bremmer 1989).

It is unwise, however, to overgeneralise such findings. The interplay between rainfall and infiltration characteristics may be entirely different in other parts of the world. For instance, Patnaik and Virdi (1962) reported much lower values for saturated infiltration rates under forest and agricultural cropping elsewhere in the Himalaya. Moreover, overland flow and surface erosion may be rampant in plantations of teak (Bell 1973; Wolterson 1979), *Gmelina* and *Shorea* (particularly when subjected to grazing and regular burning), or indeed in any forest where litter is collected to serve as fuel or compost (Wiersum 1984).

Plantation Management and Hydrology

Effects of forest thinning and coppicing

The hydrological effects of thinning forest plantations are generally small and ephemeral, unless the treatment is particularly severe (>50% of basal area removed). For instance, in 25-year-old stands of *Pinus taeda* in the southern U.S., Rogerson (1967) reported an increase in relative amounts of throughfall from 77% to 94% of incident rainfall when going from a stand density of 1556 to 124 trees ha^{-1}. Storm size alone explained almost 99% of the variation in throughfall; adding stand basal area hardly increased this value. In the tropics, Veracion and Lopez (1976) and Florido and Saplaco (1981) found negligible increases in throughfall after thinning 10–15- and 30–year-old natural forests of *Pinus kesiya* in the Philippines by 30%–50% of the biomass. Significant increases were obtained, however, when 70% was removed. Similarly, Bons (pers. comm.) observed only marginal differences in throughfall in mature plantations of *Pinus merkusii* of very different stocking (240–560 trees ha^{-1}) in Java. Ghosh et al. (1980), working in broad-leaved forest (*Shorea robusta*) in northern India, applied a 20% thinning treatment and found throughfall to increase from 72% to 81%. However, because of a concurrent reduction in stemflow (from about 10% to 6% and not necessarily related to the thinning), the overall change in intercepted rainfall was a modest 5%.

It would seem, therefore, that the effect of forest thinning on rainfall interception is much less than would be expected on a pro-rata basis. This may be due to the fact that the effect of a reduction in canopy biomass is compensated to some extent by increased ventilation, and thus reduced boundary layer resistance, of the remaining trees (Teklehaimanot et al. 1991). Also, as noted by Waterloo (1994) during a study of pre- and posthurricane throughfall in *Pinus caribaea* plantations in Fiji, the effect of reduced interception by the canopy is also counteracted (typically by 1%–2%) by a concurrent increase in the amount of water intercepted by the litter layer. Not only does the latter become more exposed after the opening up of the canopy but it also has a greater biomass capable of storing (intercepting) more moisture. Another effect of thinning may be the stimulation of growth in the understorey (cf. Roberts 1983). Thus the effect of thinning on streamflow characteristics may be even smaller than that on rainfall interception.

Gilmour (1977) did not observe any changes in streamflow after selectively logging a rainforest in northern Queensland. Subba Rao et al. (1985) reported a slight increase (by 8.6%) in peak flow rates (but not in stormflow volume) in the thinning experiment in northern India already referred to. The effect had disappeared by the second year. Under sub-humid conditions the effect of thinning on streamflow will be even smaller than in areas with adequate rainfall because the remaining trees will tend to make use of the extra moisture in the soil produced by the thinning (Bosch 1979), and thus rather drastic treatments will then be needed to achieve significantly increased streamflow (Stoneman 1993).

Coppicing is such a drastic measure. The more or less complete removal of the above-ground biomass, while leaving the root system intact, will lead to high water use by trees as they rapidly redevelop their biomass. Replacing degraded scrub with *Eucalyptus grandis* and *Eucalyptus camaldulensis* in northern India produced an average reduction in streamflow (mainly stormflow) of 26% during the first five yr (Mathur et al. 1976; Mathur and Sajwan 1978). However, when the trees were coppiced, this figure rose to 68% in the first year and to 47% in the second year. The effect had disappeared by the third year after coppicing (Viswanatham et al. 1980, 1982).

Riparian zone management

The hydrological benefits of maintaining an undisturbed riparian buffer zone range from detaining overland flow from upslope areas and trapping the sediment carried by it, to keeping streamwater temperature fluctuations within acceptable limits (Clinnick 1985). Storm runoff from undisturbed forested areas in the more humid parts of the world is generated to a large extent in the near-saturated areas around streams. During storm events these 'contributing areas' may become linked to the more ephemeral elements of the stream network that act as a source of runoff to the permanent section of the stream (Ward 1984). Therefore, it is important to extend the riparian buffer zone beyond the point where regular streamflow

commences and to include these ephemeral elements, even more so because the position of springheads may move upstream after forest clearing, reflecting the wetter conditions in the catchment as a result of decreased evaporation (Bren and Turner 1985). The importance of such an extended buffer zone was illustrated by O'Loughlin et al. (1980) in a study of sediment yields after clearcutting a hardwood forest in New Zealand.

The width of streamside buffer strips required for satisfactory stream protection has been a matter of debate (Clinnick 1985). Where runoff response is marked and water quality is at risk, riparian buffer strips may need to be fairly wide. This may be the case in areas with significant contributions from valley bottom SOF or from HOF generated on heavily disturbed side slopes (O'Loughlin et al. 1980; Fritsch 1993). However, when soils exhibit high infiltration rates, narrower buffers may suffice (Clinnick 1985). The recent development of powerful topography-based distributed hydrological models (O'Loughlin 1986; Moore et al. 1991) has improved our ability to predict the location and extent of surface saturation zones within the landscape for a range of climatic and pedological conditions. Pending the widespread application of such models in commercial forestry, many people consider widths of 10–30 m on either side of a perennial stream and of about 5 m around ephemeral channels sufficient (Clinnick 1985).

What are the associated costs of streamside protection in terms of water use by riparian vegetation, roots of which have ready access to the groundwater table? Dunford and Fletcher (1947) found under humid warm-temperate conditions in southeastern USA that removal of the riparian vegetation produced a decrease in the diurnal fluctuation of the baseflow but that the associated increase in water yield was not greater than would have been expected if an equal area elsewhere in the catchment had been cleared. Apparently, the riparian effect is negligible in areas where soil moisture remains readily available in all parts of the catchment throughout the year. A similar result was obtained for humid subtropical conditions in South Africa by Smith and Bosch (1989).

Minimising adverse impacts on soils and hydrology during forestry operations

There is ample evidence that the key to minimising damage to residual vegetation and soil is the careful planning, preparation and execution of a forestry operation.

Short-term economies may well be offset by the costs of restoring or maintaining productivity in the longer term. For instance, the growth of *Eucalyptus urophylla* and *Acacia mangium* trees during the first four yr after forest clearance on clayey soils by heavy equipment, windrowing and burning of slash in French Guyana and East Malaysia, respectively, was roughly half of that of trees on better-managed soils (Sim and Nykvist 1991; Fritsch 1992). In the Malaysian study the manual extraction of timber and retention of slash on-site to rot rather than burning it off, although initially more expensive than the conventional use of crawler tractors and burning of slash, proved to be more economic soon after planting. Not only was tree growth better, but also less weeding was required. In addition, adverse off-site effects such as increased catchment sediment yield (not included in the economic analysis) were lower after the more benign treatment (Malmer 1993).

Reductions in early growth might also be related to enhanced losses of nutrients due to leaching and surface erosion after the loss of the protective litter layer by the burn (Malmer 1993; Mackensen 1994) although the effect is hard to separate from that of soil compaction. Changes in soil infiltration capacity due to fire may range from negligible to considerable. Much depends on the intensity of the fire which is governed mainly by amounts and moisture content of the fuel and ambient weather conditions (Scott 1993; Mackensen 1994). Different soils may exhibit a different response to excess heat; some soil types become (temporarily) water repellent, a condition which tends to promote the generation of overland flow (Burch et al. 1989; Scott 1993). When this occurs on sites where most of the protective litter layer has just been destroyed by the fire, serious surface and gully erosion may follow (Brown 1972; Leitch et al. 1983).

Forest plantations in East Africa and Southeast Asia are often established under the 'taungya' system (Evans 1992). As the soil remains partly exposed throughout the intercropping period, the risk of soil degradation through erosion increases, particularly because taungya is often applied on steep slopes. In Java, freshly cleared fields on highly porous volcanic ash soils of sandy loam texture did not incur significant surface erosion during the first two yr of intercropping (Bons 1990; Rijsdijk and Bruijnzeel 1990) whereas in the second year coarser textured volcanic soils which were more prone to water repellent behaviour and structural decline did (Gonggrijp 1941). However, after the first year of taungya the former studies recorded intensive overland flow and erosion (up to 70 t ha^{-1} yr^{-1}) from compacted field boundaries used as footpaths. Even though the resulting gullies may stop supplying large amounts of sediment to the stream after the cessation of cultivation and the closing of the tree canopy, they may remain important as preferential routes for storm runoff. Under more marginal conditions, e.g. on the vertic clay soils often used for teak in Java, taungya may lead to heavy erosion, even to the extent that protective contour barriers of teak branches and *Leuceaena* hedgerows are destroyed (Wolterson 1979).

Without going into much detail here, the following steps and measures are generally considered essential for an operation to be successful in terms of minimising both costs and environmental damage (Marn and Jonkers 1981; Wiersum 1985; Pearce and Hamilton 1986; Adams and Andrus 1990):

I. Prelogging assessment of potentially unstable (wet depressions, very steep slopes) areas or erodible soil types; delineation of catchment boundaries and (ephemeral) drainage lines; evaluation of seasonal distribution of rainfall;

II. Preplanning of the extraction road and track network in relation to terrain characteristics, the natural drainage network, and type of logging system to be used; locating roads and log landings on ridges and away from streams and steep sections as much as possible; where unavoidable, constructing stream crossings of high standard, with access of machinery to the stream zone strictly confined to these designated crossings;

III. Timing of road construction to conform to the period of least rainfall; allowing sufficient time for earthworks to stabilise before intensive use; providing adequate drainage of roads/tracks; reducing tyre pressure of vehicles to reduce rutting of roads;

IV. Where economically feasible, using skyline systems rather than ground-based traction in steep terrain; using tracked rather than rubber-tyred machinery and restricting machine size in accordance with log weights and wetness of the soil; suspending tractor logging during very wet periods to avoid excessive compaction; minimising the number of vehicle passes by sound planning of timber extraction; yarding logs uphill rather than downhill; using winch ropes from the ridges rather than having machines clear an approach to every log; delimbing logs and leaving slash on site; limiting the burning of slash to avoid nutrient losses through volatilisation, surface erosion and leaching;

V. Maintaining buffer strips along streams (10–30 m wide, depending on slope steepness and soil erodibility) and ephemeral channels (>5 m wide);

VI. Postlogging rehabilitation of skidder tracks (removal of temporary stream crossings, construction of cross drains or bars on critical tracts) and log landings (ripping and seeding); road maintenance; grassing roadside earthworks;

VII. Between harvesting periods, maintaining an adequate ground cover to protect the soil against the enhanced erosivity of canopy drip, and limiting the collection of understorey vegetation, branches and litter for fodder, fuel or composting material.

These principles should be applied through comprehensive guidelines. Although there is a need for site-specific guidelines to take full account of local variation in climatic and soil conditions, as an interim measure tropical forest managers might do well to follow the protective guidelines developed for the extreme climatic conditions in the coastal rainforest zone of Queensland and which have proved their value (Cassells et al. 1984).

The Role of Hydrological Modelling in Plantation Management

Hydrological models may prove useful within plantation management with respect to both on-site and off-site aspects. Examples of the former include the identification of excessively wet areas, or hillslope sections having a high surface erosion or landslide hazard. Off-site aspects relate to downstream hydrological impacts of forestry operations. The close linkage between the two is demonstrated by the numerous conflicts of interest between on-site demands for water for timber production and off-site needs for irrigation, industry and urban water supply, both in the tropics (Pereira 1989) and elsewhere (Langford 1976; Calder 1990). To resolve such conflicts, sound predictions are needed of the total water use of plantations at various stages of growth, on the basis of which rational land-use decisions may then be taken (Bosch 1982).

Traditionally, foresters have relied on time-consuming paired catchment experiments to evaluate any changes in water use after plantation establishment (cf. Fig. 5.7). Whilst this approach enabled the construction of nomograms from which changes in streamflow could be read as a function of stand age and rainfall or catchment exposure in certain areas (Nänni 1970; Douglass and Swank 1975), the results were often so variable as to render their applicability for more detailed water resources planning rather limited. Curves 2–4 in Figure 5.7 illustrate the considerable variation in the change in water yield after planting scrub and grassland to pines in South Africa, both between locations and years (curve no. 2). Afforestation with eucalypts produced yet another response (curve no. 1 in Fig. 5.7). The simple 'black-box' approach of the paired catchment technique is unable to evaluate the relative importance of the respective factors underlying such differences, and this severely limits the possibilities for extrapolation of results to other areas or periods.

During the last decade considerable advances have been made in the measurement and modelling of forest hydrological and ecological processes at the local to regional scale (Running and Coughlan 1988; Shuttleworth 1988; Hatton et al. 1992; Band et al. 1993). Particularly after the arrival of dynamic physically-based and spatially distributed catchment hydrological models (O'Loughlin 1986; Moore et al. 1991), the adequate simulation of the effect on water yield of changes in catchment vegetation without the use of ample parameter optimisations has come a significant step closer. Arguably, one of the more promising model packages is the topography-driven TOPOG modelling framework whose application, amongst others, has included the successful simulation of the long-term changes in tree growth and water yield during regeneration after clearfelling a *Eucalyptus regnans* forest (Vertessy et al. 1993, 1996). However, outputs from such models are sensitive to variations in leaf area index, the rainfall interception

coefficient and maximum canopy conductance, and soil hydraulic conductivity (Vertessy et al. 1993, 1996). This immediately points to the greatest problem associated with the application of advanced simulation models in the tropics, i.e. a lack of adequate input data. As a result, it is not possible at present to make reliable predictions of the effect of tropical plantations on annual and seasonal water yield in terms of tree species and age in relation to climatic and edaphic factors. However, despite the current lack of information there is reason for optimism in that the methodology to fill the major gaps in knowledge is well-established.

Pending application of the more sophisticated physically-based models to predict off-site hydrological impacts of forestry operations, tropical plantation managers will have to resort to less data-demanding yet reasonably process-based models such as ACRU (Schulze and George 1987) or PRMS (Lüllwitz and Flögel 1993). Nevertheless, there is scope for the use of the topography-driven distributed models like TOPOG for several important on-site applications that require little more input than a good contour map and basic soils information. These include the prediction of saturated zones in the landscape (O'Loughlin 1986); the spatial distribution of sheet erosion, gully erosion and landslide hazards (Vertessy et al. 1990; Dietrich et al. 1992; Constantini et al. 1993); and steady-state soil moisture distributions during wet and dry periods (Moore et al. 1988).

Synthesis

When describing the changes in hydrological behaviour of plantations at various stages of growth it is helpful to distinguish between an 'establishment phase' and a 'maturation phase'.

Upon establishing a new plantation, the soil is usually exposed for several months until a protective understorey redevelops, and during this period catchment water and particularly sediment yields, as well as peak flows, will be much increased compared to values observed previously under natural forest. The photographs on pages 159 and 160 illustrate the interaction between management and tree cover on hydrological processes under a range of forest conditions. The increases in sediment yield and peak discharges largely disappear within two to three yr, after which they assume more or less stable levels that are slightly above the original values. The increase in catchment water yield tends to last somewhat longer, usually until the canopy of the plantation closes, but information is scarce in this respect.

Relative amounts of rainfall interception after canopy closure differ between species as a function of canopy geometry and climatic conditions. Broad-leaved hardwoods such as teak and mahogany typically intercept about 20% of the rain whereas the light crowns of eucalypts intercept about 12%. Reported values for fast-growing stands of *Acacia mangium* are particularly high (up to almost 40%) whereas those for conifers are variable, but generally well under 25%. Little is known of the water use of many plantation species, particularly hardwoods such as teak, mahogany, *Acacia, Gmelina, Albizia* and *Terminalia*. To a lesser extent, this also holds true for coniferous species and (young) eucalypts. As a result, it is not possible at this stage to make reliable predictions of the effect of plantations, as the stands mature, on annual and seasonal water yield. However, there is no evidence as yet to suggest that the water use of plantations exceeds that of natural forest. On the other hand, there is increasing evidence that the planting of fast-growing trees on grasslands will lead to greatly diminished streamflow after canopy closure, particularly during the dry season. Increases in annual water use exceeding 500 mm have been reported after afforesting (sub)tropical grassland.

Despite the current lack of sound information on plantation water use there is reason for optimism in that the methodology (in terms of equipment and physically-based models) required to fill the major gaps in knowledge is well established. It is proposed that future research efforts be concentrated on the main species used in tropical forest plantations (Evans 1992) and at a relatively small number of well-researched key locations. These could be linked together in a network that captures the main features of the environmental variability encountered in the humid tropics (Bruijnzeel and Rahim 1992).

Acknowledgments

Kees Bons, Anders Malmer, Rob Vertessy and Maarten Waterloo shared unpublished data or papers. The manuscript benefited from comments offered by Drs Joe Landsberg and John Roberts. Marjolein Duifs-Eshuis assisted with the typing of Tables 5.1 and 5.2. It is a pleasure to thank Irene Sieverding for support throughout the writing of this paper and Mr H. Sion for help in preparing the figures.

Heavy erosion is often observed on poorly-drained roads, especially where erodible sub-soil is exposed in steep road sections. (Photo: T.C. Whitmore)

Where heavy machinery is employed during logging, up to 30% of the surface may be bared in roads, log landings and skidder tracks. (Photo: L.A. Bruijnzeel)

Plantations may afford excellent protection of the surface against erosion. (Photo: L.A. Bruijnzeel)

Not all forest plantations provide equally good protection of the soil against erosion. (Photo: L.A. Bruijnzeel)

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