
25 The hydrological and soil impacts of forestation in the tropics¹

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INTRODUCTION

In response to the continuing degradation and disappearance of the world's tropical forests (Drigo, this volume) the establishment of plantation forest on degraded and previously forested sites as well as into (sub)tropical grasslands is becoming increasingly common (Evans, 1999). The hydrological effects of this practice and the potential of forestation to improve or restore the hydrological behaviour of degraded catchments constitute the prime focus of this chapter, expanding and updating an earlier review of the subject by Bruijnzeel (1997). Three aspects are highlighted in particular, namely: (i) the effects of tree plantations on annual and seasonal streamflow totals; (ii) the associated impacts on stormflow and sediment production; and (iii) concurrent changes in soil chemical characteristics (fertility). Because the hydrological changes associated with forest clearing and the establishment of a new vegetation cover during the first few years are discussed at length in the chapter by Grip, Fritsch and Bruijnzeel, much of what follows pertains to the post-canopy closure phase of plantations.

EXTENT, DEVELOPMENT AND IMPORTANCE OF TROPICAL TREE PLANTATIONS

The establishment of timber plantations is a notable and accelerating land-use development of the last half-century. It has been estimated that there are now some 40 to 50 million ha of forest plantations in the tropics and warmer subtropics, trees being planted nowadays at a rate of *c.* 2 million ha yr⁻¹ compared

to *c.* 1 million ha yr⁻¹ a decade ago (Evans, 1999). One of the most widely used types of tree is the eucalypt (*Eucalyptus* spp.). Eucalypt plantations cover more than 17 million ha world-wide (Table 25.1) (FAO, 2001). More than 90% of these have been established since 1955 and roughly 50% during the last decade (Turnbull, 1999).

Despite the increase in rates of tropical plantation establishment in recent years, the rate of closed forest destruction far exceeds that of plantation establishment. Evans (1986) estimated a ratio of 11:1 between these two early on, whereas the data presented by Drigo (this volume) suggest even higher ratios (18–24:1). However, the latter estimates are based on remote sensing surveys in which plantations with poor establishment and low cover caused by weak growth are often missed (R. Drigo, pers. comm.). Based on pan-tropical remote sensing surveys, between 1980 and 1990 most (visible) new plantations were raised on sites that originally supported natural forest whereas between 1990 and 2000 more than half of the new plantations were established on other land cover types, including grasslands (degraded or otherwise), and, to a lesser extent, savannahs (Drigo, this volume). A distinction is sometimes made between plantations in the dry tropics (mean annual precipitation <800 mm; rainy season less than 4.5 months) and the more humid parts of the tropics and subtropics having rainy seasons of variable length (cf. Chang and Lau, 1993). Similarly, a distinction can be made between low-lying and high altitude areas, of which those in humid

¹ Strictly speaking, the term 'afforestation' should be used for the planting of trees in areas where forest is naturally absent, as opposed to 'reforestation' for previously forested, and subsequently degraded areas. To avoid semantic problems, the term 'forestation' (Wiersum, 1984a) is used mostly throughout this chapter.

Table 25.1. *The area of plantations around the globe by region and type*

Region	Area		Plantation areas by genus groups (10 ³ ha)							
	10 ³ ha	Percent of total	<i>Acacia</i>	<i>Eucalyptus</i>	<i>Hevea</i>	<i>Tectona</i>	Other broadleaf	<i>Pinus</i>	Other conifer	Unspecified
Africa	8 036	4.3	345	1 799	573	207	902	1 648	578	1 985
Asia	115 847	61.9	7 964	10 994	9 058	5 409	31 556	15 532	19 968	15 365
Europe	32 015	17.1	–	–	–	–	15	–	–	32 000
North and Central America	17 533	9.4	–	198	52	76	383	15 440	88	1 297
Oceania	3 201	1.7	8	33	20	7	101	73	10	2 948
South America	10 455	5.6	–	4 836	183	18	599	4 699	98	23
World total	187 086	100.0	8 317	17 860	9 885	5 716	33 556	37 391	20 743	53 618

Source: FAO (2001).

tropical uplands are commonly thought to be the most productive (Evans, 1999).

Clearly, the various types of plantations fulfil significant economic and ecological functions. Usually, a distinction is made between industrial and non-industrial forest plantations (FAO, 1995). Non-industrial plantations supply fuelwood and construction timber, charcoal, fodder, shelter and non-wood products for local consumers on a small scale, while industrial plantations are established for the larger-scale production of pulpwood, sawlogs and veneer. The area of industrial plantations is often small compared to the area of native forest, but their share in wood production is usually highly significant. For example, plantations in Brazil, Zimbabwe or Zambia each represent <1.3% of the national forest area but they contribute 50–60% of the wood production (FAO, 1995). This is because industrial plantations generally exhibit high productivity and are readily manageable. Especially fast-growing tree plantations using eucalypts, pines or acacias in short rotations of between 10–30 years are favoured in this regard (Brown *et al.*, 1997). In South Africa, by way of example, roughly 95% of all the country's timber needs are provided by productive and intensively managed plantations of pine, eucalypts and wattle (mainly *Acacia mearnsii*) covering just 1.5 million ha (1.2% of the land area), while wood exports contributed a further US\$1.01 billion in foreign exchange earnings in 1998 (R. Godsmark, pers. comm.). Besides their role for wood production, plantations are also valued for environmental benefits such as soil protection against erosion and rehabilitation of degraded land, for diverting pressure off natural forest and, in some cases, the enhancement of biodiversity (Evans, 1999).

It is difficult to find reliable figures on the areas of plantation established on previously degraded land as opposed to landscapes in good hydrological condition, but it can be assumed that the largest growth is found in the area of essentially commercial plantations, where there is a strong profit motive, rather than in the

establishment of plantations for broader social or environmental reasons. As productivity is related to the quality of the site, most industrial plantations can be expected to be on sites and soils that were not degraded. In South Africa, for example, large areas of the humid grassland regions are degraded due to over-grazing, and these lands are specifically avoided by private industrial forestry companies because of their low productivity.

HYDROLOGICAL IMPACTS OF FORESTATION

To predict the hydrological effects of a land cover change, it is important to understand what the treatment involves: is it just a change in vegetation cover, or are there associated changes to important soil properties as well? For example, deforestation followed by heavy grazing or intensive cropping accompanied by marked changes in soil physical properties (notably bulk density and infiltration capacity; Lal, 1987; Godsey and Elsenbeer, 2002) should be viewed differently from deforestation followed by the development of secondary forest with comparatively little change in vegetation leaf biomass and soil structure (Giambelluca, 2002; Hölscher, Mackensen and Roberts, this volume). Similarly, the establishment of forest in catchments in good hydrological condition (i.e. where soil infiltration capacities have been maintained) is fundamentally different from forestation of truly degraded catchments. Before reviewing the effects of forestation on precipitation, annual and seasonal water yields, storm runoff and sediment production, the water use of tropical tree plantations will be compared with those of the rainforests, grasslands and agricultural crops they may be replacing. Effects of forestation of severely degraded catchments will be treated separately in view of their different hydrological behaviour.

Water use of tropical tree plantations

Compared to short vegetation types like grass, scrub or agricultural crops, forests have greater leaf area and canopy height. Because of this their aerodynamic roughness and capacity to intercept rainfall are enhanced; in addition, forests absorb more radiation (typically by 5–10%) and have deeper root systems. The latter enables them to maintain transpiration during dry spells when more shallow-rooted plants are likely to experience water stress and thus are forced to reduce their water uptake (cf. Roberts *et al.*, this volume). The contrast is particularly pronounced in the seasonal (sub)tropics where grasslands and sclerophyllous scrub become dormant during the extended dry season (Waterloo *et al.*, 1999; Smith and Scott, 1992) and crop cover is much reduced or fields lie bare (Harding *et al.*, 1992; Van Dijk, 2002). As such, establishing tree plantations on grass-, scrub- or crop lands whose soils have not become degraded can be expected to lead to more or less serious reductions in soil moisture and, ultimately, groundwater recharge and water yield (Hamilton and King, 1983; Bruijnzeel, 1990).

Total evapotranspiration (E_T) of a forest consists of evaporation from a wet canopy (rainfall interception, E_i), evaporation from a dry canopy (transpiration, E_t), and that from the forest floor (soil or litter evaporation, E_s). E_s is generally considered to be small under fully closed canopies, although it cannot be neglected under more open conditions (Waterloo *et al.*, 1999; Putuhena and Cordery, 1995; Ashby, 1999). Bruijnzeel (1997) reviewed the results of some 20 rainfall interception studies in humid tropical tree plantations for which comparatively good data are available. Eucalypts generally exhibit low values (c. 12%), broad-leaved hardwood species such as teak and mahogany typically intercept about 20%, whereas pines and other conifers (*Araucaria*, *Cupressus*) mostly fall in the range of 20–25%, with the higher values usually found in upland situations. Well-developed stands of the particularly fast-growing *Acacia mangium*, on the other hand, exhibited values of 20–40%. Rainfall interception by another fast grower, *Paraserianthes falcataria* (a.k.a. *Albizia*), which is increasingly planted in South East Asia for soil rehabilitation purposes (Van Dijk and Bruijnzeel, 2003), typically reaches 18–20%, reflecting its much lighter canopy compared to *A. mangium*. Typical values for the rainforests replaced by these plantations range from 10 to 20% in most lowland situations to 20–35% in montane forests (Bruijnzeel, 1989a; Bruijnzeel and Proctor, 1995).

The database for *annual* estimates of tropical tree plantation water use (E_T , E_i) is very limited. Table 25.2 (modified and updated from Bruijnzeel, 1997) summarises some key results. To facilitate comparisons between species and locations, values for ET in Table 25.2 have been normalised by dividing them by the corresponding open water evaporation (Penman's E_o) wherever

such information was available. Corresponding estimates for natural forest or grassland in some of the locations have been added to illustrate the contrast or similarity in water use by the respective vegetation types.

The data in Table 25.2 are of variable quality (see footnotes). In many cases E_T has been estimated from the catchment water budget and may have been overestimated somewhat because of potential catchment leakage (cf. Roberts *et al.*, this volume). Similarly, in several cases E_i has been derived by subtracting interception losses from E_T , thereby introducing additional uncertainty, whereas in others annual totals were estimated by simple extrapolation of short-term measurements (e.g. for *A. mangium*). Despite these uncertainties, the data permit a few tentative conclusions:

- (1) Water use by mature tropical plantations of average stocking resembles that of old-growth forest in the same area (East Malaysia, Indonesia, Jamaica, Kenya) and possibly exceeds it in the case of particularly vigorous growth (*A. mangium* in East Malaysia, *Pinus caribaea* in Fiji).
- (2) Interception losses constitute a significant portion of overall E_T , particularly in humid tropical upland areas.
- (3) Very high transpiration totals are found in maritime tropical situations (Fiji, Jamaica, Atlantic lowlands of Costa Rica).
- (4) None of the forests listed in Table 25.2 seem to be subject to serious soil water stress.
- (5) Plantation water use invariably exceeds that of rain-fed crops, pasture or fire-climax grassland in the same area, particularly under more seasonal conditions (Fiji, Tanzania).

Additional modelling-supported process-based work along the lines of Waterloo *et al.* (1999), Ciencala, Kuczera and Malmer (2000) or Bigelow (2001) is needed to obtain more reliable estimates and test some of these conclusions. However, whilst plausible estimates of E_i appear feasible from short-term measurements of stomatal behaviour made under representative climatic conditions and using the Penman-Monteith model of evaporation, a similar approach for wet canopy evaporation seems to produce less reliable results. For example, the interception estimates derived from short-term observations and subsequent modelling for various young stands in Costa Rica by Bigelow (2001) seem distinctly low (Table 25.2), possibly because advective effects were not incorporated. Waterloo *et al.* (1999) reported similar discrepancies between measured and modelled E_i for pine stands in Fiji (cf. discussion in Roberts *et al.*, this volume).

In view of its widespread use, the almost complete absence of reliable estimates of annual water use by *Eucalyptus sp.* under humid tropical conditions is striking, although such data are available for more subhumid tropical conditions, as well as from subtropical and warm-temperate areas. Examples include Australia (reviewed by Vertessy *et al.*, 2003), south-east Brazil (Lima *et al.*, 1990), South Africa (Scott and Smith, 1997; see also

Table 25.2. Estimates of annual evapotranspiration (E_T), interception (E_i) and transpiration (E_t) for tree plantations in the humid tropics after canopy closure (values rounded off to the nearest 5 mm).

Species	Location	Elevation (m)	Mean annual precipitation (mm)	Age (yr)	E_T (mm)	E_T/E_0	E_i	E_t
							(mm)	
<i>Acacia mangium</i> ^a	Sabah	700	3280	9–10	1495	>1.0	655	840
<i>Agathis dammara</i> ^b	Java	600	4770	11–35	1070	0.79	665	405
<i>Cedrela odorata</i> ^c	Costa Rica	100	4210	4–5	1320	–	75	1245
<i>Cordia alliodora</i> ^c	Costa Rica	100	4210	4–5	1385	–	280	1110
<i>Eucalyptus robusta</i> ^d	Madagascar	1010	2000	>50	1505	–	–	–
Montane rainforest ^d	Madagascar	1010			1295	–	–	–
<i>Hyeronyma alch.</i> ^c	Costa Rica	100	4210	4–5	1510	–	375	1135
<i>Pinus caribaea</i> ^e	Fiji	80	1800	6	1925	1.13	540	1385
	Fiji	230		15	1715	1.01	495	1220
Fire-climax grassland ^{e,f}	Fiji	90			750	0.47	195	555
<i>P. caribaea</i> ^g	Jamaica	1020	3745	19	1850	–	635	1215
<i>P. elliotii</i> ^h	Queensland	60	1260	>35	1082	0.76 ⁺	–	–
<i>P. merkusii</i> ⁱ	Java	1300	2100	31	900	0.84	555	445
<i>P. patula</i> ^k	Kenya	2400	2305	>10	1160	0.77	560	600
Montane rainforest ^k	Kenya	2440			1155	0.78	465	690
Crops and pine seedlings ^k	Kenya	2400		<3	1030	0.69	–	–
Crops and grassland ^l	Tanzania	2500	1925		970	0.65	–	–
Lowland rainforest ^m	Manaus, Brazil		2075		1315	0.77	335	980
Pasture ⁿ	Manaus, Brazil		2065		915	0.54	125°	790°
Lowland rainforest ^p	Java		2850		1480	0.90	595	885

^a Cienciala *et al.* (2000): approximate annual E_t derived by extrapolation of short-term sapflow measurements in stand of average basal area; E_i based on pers. comm. by A. Malmer; E_T of rainforest c.1835 mm (Malmer, 1992).

^b Bruijnzeel (1988): catchment water balance (CWB), E_t by subtracting E_i from E_T and hence approximate only.

^c Bigelow (2001); E_t modelled with Penman–Monteith equation, E_i derived with Rutter model and possibly seriously underestimated.

^d Bailly *et al.*, 1974: CWB, some leakage possible; E_T for lower montane rainforest (LMRF) c.1295 mm.

^e Waterloo *et al.* (1999): E_t via micro-meteorological methods, E_i including c.160 mm evaporated from litter layer.

^f Waterloo *et al.* (in press): methods as in (^e).

^g Richardson (1982): CWB, leakage possible; E_T for LMRF c.2000 mm.

^h Bubb and Croton (2001): CWB, ⁺pan evaporation.

ⁱ C. A. Bons, pers. comm.

^k Blackie (1979): CWB, E_i based on Pereira (1952).

^l Edwards (1979): CWB, long dry season.

^m Shuttleworth (1988): methods as in (^c, ^e).

ⁿ Kabat *et al.* (1999).

^o Ashby (1999): as in (^c, ^e).

^p Calder *et al.* (1986): as in (^c).

below), south India (Harding *et al.*, 1992; Calder *et al.*, 1997), and Pakistan (Morris *et al.*, 1999). The planting of eucalypts in seasonal climates has met particularly vigorous opposition in the popular environmental literature, mainly because they are thought to be ‘voracious consumers of water’ (e.g. Vandana Shiva and Bandyopadhyay, 1983). Indeed, young plantations of *Eucalyptus camaldulensis* and *E. tereticornis* in south India exhibited such behaviour with transpiration rates of up to 6 mm day⁻¹ when unrestricted by soil water deficits at the end of the monsoon, although

values fell to 1 mm day⁻¹ during the subsequent long dry season (Roberts and Rosier, 1993). Because of this stomatal regulation, the annual water use of the plantations on soils of intermediate depth (c. 3 m) was not significantly different from that of indigenous dry deciduous forest (Calder *et al.*, 1992; Harding *et al.*, 1992). However, on much deeper soils (>8 m) eucalypt water use exceeded annual rainfall considerably, suggesting ‘mining’ of soil water reserves that had accumulated previously in deeper layers during years of above-average rainfall. Moreover, the rate

of root penetration was shown to be at least as rapid as 2.5 m yr^{-1} and roughly equalled above-ground increases in height (Calder *et al.*, 1997). Similar observations have been made for *E. grandis* in South Africa (Dye, 1996; see also below). It is probably pertinent in this respect that Viswanatham *et al.* (1982) observed strong decreases in streamflow after coppicing of *E. camaldulensis* in northern India. A recent catchment experiment involving *E. globulus* in south India (Sharda *et al.*, 1998) confirmed the enhancing effect of coppicing: the reduction in water yield during the second rotation of 10 years (first generation coppice) was substantially higher (by 156%) compared to that observed during the first rotation although the roots did not reach the groundwater table (Samraj *et al.*, 1988). All of the above findings confirm the original fears of Vandana Shiva and Bandyopadhyay (1983) although Collopy *et al.* (2000) recently reported surprisingly conservative water use by a highly productive *E. urophylla* plantation on the Leizhou Peninsula in China under conditions of low vapour pressure deficit. In summary, planting of eucalypts, particularly in sub-humid climates, should be based on judicious planning, i.e. away from water courses and depressions or where the roots would have rapid access to groundwater reserves.

Similar caution may have to be applied in the case of *Acacia mangium*, another tree capable of very high growth rates (Lim, 1988). Although the extrapolated annual E_T and E_r values for an *A. mangium* stand of average growth and stocking in East Malaysia were not excessive (Table 25.2), average transpiration rates in a denser and more productive stand were 170% higher and showed no consistent stomatal control in response to changes in atmospheric vapour pressure or soil water deficits even though the measurements were made during a relatively dry part of the year (Cienciala *et al.*, 2000). Since the rainfall interception fraction of the more productive stand was also higher (28% vs. 20% in the average stand; A. Malmer, pers. comm.) it cannot be excluded that total annual water use for these vigorously growing trees exceeds that of the forests they are replacing by at least several hundreds of mm (cf. Malmer, 1992). Further work is needed to test this contention.

Effect of forestation on precipitation

Because tree plantations share many of the vegetation characteristics considered to be responsible for the high evaporation from tropical rainforests (such as high leaf area index, low albedo, high aerodynamic roughness and deep roots; Roberts *et al.*, this volume), the establishment of extensive fast-growing plantations over large areas with degraded vegetation may be expected to exert a positive influence on the moisture content of the atmospheric boundary layer (ABL), and (by implication) on cloud formation and rainfall (Dolman *et al.*, 2004). Direct observational evidence to test this contention appears to be lacking for plantation forest, but both physical reasoning and model simulations indicate that

large-scale forest conversion to pasture (i.e. the reverse of forestation) may cause moderate decreases in rainfall (as reviewed by Costa, this volume). Again, verification of corresponding long-term changes in rainfall in areas subject to major land use change has proved difficult (Van Rompaey, 1993; Wilk, Andersson and Plermkamom, 2001). This is because of natural climate variability, uncertainty about the relative importance of atmosphere-ocean interactions and regional surface characteristics, phase lags between rainfall and vegetation development, and (not least) uncertainties in rainfall and vegetation data (Zeng *et al.*, 1999; Bruijnzeel, 2004). Nevertheless, there is observational evidence that forest conversion to pasture over areas between 1000 and 10 000 km^2 has caused changes in the timing and distribution of clouds in parts of Amazonia (Cutrim *et al.*, 1995) and Costa Rica (Lawton *et al.*, 2001).

To explore the potential effects of large plantations of *Eucalyptus camaldulensis* on the regional climate of southern India (Karnataka), Harding (1992) used a simple one-dimensional ABL model. The simulations were limited to the post-monsoon season (December) for which previous work had shown very low evaporation rates for crop land ($<1 \text{ mm day}^{-1}$) but much higher values for the eucalypts ($3\text{--}4 \text{ mm day}^{-1}$). Despite its simplicity, the model was able to reproduce the daily maximum height of the ABL to within the resolution of measured radiosonde data. In addition, the model reproduced the broad features of the observed daily variation in surface temperature and humidity. Next, the model was used to predict the temperature and moisture content of the ABL associated with the two land cover types. The ABL forming above the eucalypt plantation was predicted to be 3.7°C cooler and 0.4 g kg^{-1} moister than that above the crops, implying a specific humidity deficit above the forest that was reduced to 57% of the value predicted for bare soil.

Furthermore, the model was used to estimate the minimum size of a forest that would have a measurable effect on atmospheric humidity. A 10 km stretch of forest was shown to have a small effect on humidity which was measurable but not large enough to have a significant effect in terms of feedback to atmospheric evaporation demands downwind of the forest. Conversely, forests in excess of 50 km were predicted to have a significant effect on downwind evaporation rates. It should be noted, however, that the one-dimensional nature of the model used by Harding is likely to have overestimated the effect of changes in the vegetation because three-dimensional circulations, which tend to increase horizontal mixing of the air, were not incorporated (cf. Costa, this volume). In addition, only the feedback between evaporation and evaporative demand of the atmosphere could be addressed by the model, but not the effect on rainfall. This would require the use of a full three-dimensional meso-scale circulation model (such as RAMS; Pielke *et al.*, 1992) and realistic surface parameterisation (Lawton *et al.*, 2001; Van der Molen, 2002; Dolman *et al.*, 2004). The

information on vegetation characteristics required for such simulations is now gradually becoming available for tropical grasslands (Wright *et al.*, 1996; Waterloo *et al.*, 1999, in press), various types of tree plantations (Beadle, 1997; Waterloo *et al.*, 1999; Bigelow, 2001) as well as rain-fed upland crops (Van Dijk and Bruijnzeel, 2001; Van Dijk, 2002). However, similar information on concurrent changes in soil physical properties (needed for the adequate representation of soil water feedbacks on tree water uptake in the model) is still scarce (Lal, 1987; see also section on forestation of degraded land below).

Effects of forestation on water yield

In view of the contrasts in water use between forests and shorter vegetation types discussed in the previous sections (Table 25.2) it is not surprising that the establishment of timber plantations into grassland or scrub leads to an increase in evaporative losses and a resultant decrease in *annual* streamflow totals. Although no stringent (paired) catchment studies have been conducted to actually demonstrate such reductions in flow upon forestation in the humid tropics proper, there is overwhelming evidence to this extent from the subhumid tropics (Samraj *et al.*, 1988; Sikka *et al.*, 2003), the subtropics (Scott and Smith, 1997; Scott *et al.*, 1999) and the temperate zone (Bosch and Hewlett, 1982; Trimble *et al.*, 1987; Fahey and Jackson, 1997). By analogy, there is a dataset comprising more than 100 long-term, controlled paired catchment experiments showing unequivocally that the felling of native forest or timber plantations (i.e. the reverse of forestation) under a broad range of climatic and topographic conditions results in *increases* in catchment water yield. The increase is positively related to the proportion of the catchment that is affected or the proportion of biomass removed (Bosch and Hewlett, 1982; Sahin and Hall 1996; Stednick, 1996; for the tropics: Bruijnzeel, 1990, 1996).

Reported initial increases in water yield following the clear-felling of forest, in both tropical and temperate regions, generally range between 25–60 mm yr⁻¹ per 10% of catchment affected. However, maximum increases of 80–90 mm yr⁻¹ per 10% forest removal have been recorded in some tropical studies (Bruijnzeel, 1990; Grip *et al.*, this volume). The observed variation in initial hydrological response to clearing is considerable and can be explained only partially by differences in rainfall between locations or years (Bruijnzeel, 1996). Other factors include differences in elevation and distance to the coast (affecting evaporation; Roberts *et al.*, this volume); catchment steepness, soil depth and changes in permeability with depth (governing the residence time of the water, speed of baseflow recession and stormflow generation patterns; Bonell, this volume); and, above all, the degree of disturbance of undergrowth and topsoil by machinery or fire (determining both infiltration and rate of regrowth; Grip *et al.*, this

volume; Hölscher *et al.*, this volume). Because the relative importance of the respective factors varies between sites, additional process studies are usually required if the results of paired basin experiments (which essentially represent a black box approach) are to be fully understood and extrapolated to other areas (Bruijnzeel, 1990, 1996; Bonell and Balek, 1993). It is pertinent to note that the bulk of the increase in flow is usually observed in the form of baseflow rather than as greatly increased stormflows. This appears to be in contrast to the deterioration in flow regime that is so often observed in practice following tropical forest conversion to grazing or cropping (Pereira, 1989). Indeed, it may reflect a lack of realism on the part of most experimental studies in which either soil disturbance upon clearance remained limited or post-forest land use did not last long enough to degrade the soil sufficiently (Bruijnzeel, 1996; Sandström, 1998).

Whether the increases in water yield following forest clearing are temporary or permanent depends on the type of the new vegetation. In the case of natural regeneration, streamflows can be expected to revert to pre-clearance values within a decade (Giambelluca, 2002; Hölscher *et al.*, this volume), whereas forest conversion to pasture or annual cropping produces permanent increases (Grip *et al.*, this volume) (Table 25.2). Similarly, in one of the few examples where tropical tree plantations replaced natural forest, water yield returned to original levels after 6–10 years in the case of *Pinus patula* in a montane area with deep volcanic soils in Kenya (Blackie, 1979). An analysis of the changes in flow during the first seven years after converting logged-over rainforest to vigorously growing *Acacia mangium* in East Malaysia revealed a drop below pre-clearing streamflow levels from the fourth year onwards (cf. Figure 21.2 in Malmer *et al.*, this volume). Such rapid reductions undoubtedly reflect the high transpiration and interception of *A. mangium* cited earlier (cf. Ciencala *et al.*, 2000).

Waterloo *et al.* (1999) investigated the hydrological effects of reforesting fire-climax grassland with vigorously growing stands of *P. caribaea* in Fiji (MAP 2000 mm, dry season of 5 months) following earlier reports by Kammer and Raj (1979) of strongly diminished low flows some years after planting. Combining micro-meteorological and soil water balance techniques with modelling, Waterloo *et al.* (1999) derived (very high) annual water use totals for six- to 15-year-old pine stands (1717–1926 mm) (Table 25.2) which far exceeded evapotranspiration from the seasonally dormant grassland (c. 748 mm yr⁻¹). Because infiltration capacities of the (ungrazed) grassland soils had remained high (Waterloo, 1994), no change in the generation of overland flow after forestation was expected and any changes in streamflow would thus largely reflect contrasts in water use between the two vegetation types. Despite the massive difference in annual E_T between pines and grassland at the site level (Table 25.2), Waterloo *et al.* (1999) suggested that actual reductions in annual streamflow would be rather in the order of 500–700 mm. Reasons

for this include the fact that the pine stands had a much poorer stocking at the catchment scale due to repeated disturbance by hurricanes and the fact that indigenous forest still lined the riparian zones of grassland catchments. This riparian forest was expected to transpire at potential, if not higher rates because of positive heat advection from the warmer and drier surrounding grasslands (cf. Giambelluca, 2002). In fact, it cannot be excluded that the very high E_T totals derived for the pines themselves may be partly due to a similar mechanism (M. J. Waterloo, pers. comm.).

The planting of *Eucalyptus globulus* over 59% of a montane grassland catchment with swampy forested valley bottoms in sub-humid south India (MAP 1380 mm) did not produce a noticeable decline in streamflow during the first three years but from then onwards annual water yield declined to 120 mm below original values until the trees were coppiced at 10 years of age. This represented a reduction in flow by 21% between age 3–10 and 16% over the entire rotation (Samraj *et al.*, 1988). During a second, 10-year rotation (first-rotation coppice), flows were reduced from the beginning because the root network had remained intact. As a result, flows were reduced even more during the second rotation (by 25.4% overall; Sharda *et al.*, 1998). The increased water use during the coppice phase corresponded with increased production of woody biomass (14 t ha⁻¹ yr⁻¹ vs. 10 t ha⁻¹ yr⁻¹ during the first rotation) and a slight increase in root penetration depth (from an average 2.8 m at the end of the first rotation to 3.2 m at the end of the second; Sikka *et al.*, 2003). Such reductions in flow may seem modest compared to the 500–700 mm yr⁻¹ inferred earlier by Waterloo *et al.* (1999). However, not only does the Indian work represent a partial conversion over less than 60% of the catchment but also both rainfall and evaporation in the Nilgiris are lower than in lowland Fiji.

Case study: effects of afforestation of subtropical grasslands in South Africa on water yield

In addition to these few tropical examples, much can be learned from a particularly comprehensive series of long-term (since the 1930s) paired catchment experiments of the hydrological effects of afforesting natural grasslands, sclerophyllous shrub and evergreen, broad-leaved forest in subtropical South Africa (summarised at various stages by Bosch, 1982; Scott and Smith, 1997; Scott *et al.*, 1999). Of particular relevance here are four experiments where grasslands in good condition were afforested, one with *Eucalyptus grandis* and three with *Pinus patula*, plus one experiment where native evergreen forest was replaced with *Eucalyptus grandis*. The research sites are all in the high rainfall zone of South Africa (MAP between 1100–1600 mm). Experimental control was provided by catchments kept under native vegetation. The results of the planting experiments are summarised in Table 25.3, and illustrate the range and variability of the key criteria used

to express the effect of forestation. To facilitate comparisons the results have been standardised to a 10% level of planting by assuming a linear relationship to the area that is treated. It is important to note that these experimental catchments were all in good condition, with no or little soil erosion. Although generally steep, they have deep, well-drained soils and very low storm response ratios (Hewlett and Bosch, 1984). Therefore the experimental comparison is between the two vegetation covers (i.e. differences in total evaporation only), and does not involve the effects of concurrent changes in soil condition as would be expected upon forestation of degraded catchments (see below).

The resulting streamflow reductions over time after planting follow a sigmoidal pattern comparable to a growth curve (Figure 25.1). There are clear differences between the effects of eucalypts and pines, but there is also a large amount of variation from year to year within a single experiment and between different experiments, even in comparable catchments in one locality (see discussions in Bosch (1982), Smith and Scott (1997) and Scott *et al.* (1999) for details). The highest flow reductions occur while the tree crop is mature and range, for a 10% level of planting, from 17.3 mm or 10% yr⁻¹ in a drier catchment to 67.1 mm and 6.6% yr⁻¹ in wetter catchments (Table 25.3). The former are similar to the results obtained in South India (c. 20 mm per 10% forest yr⁻¹) whereas the latter rather resemble the results obtained in Fiji (50–60 mm per 10% yr⁻¹).

The timing of the first significant reductions in flow after planting varies quite widely depending on the rate at which catchments are dominated by the plantation crop. The pine plantations in the high altitude grasslands at Cathedral Peak usually took several years to have a clear impact on streamflow (up to 8 years) (Table 25.3, column 5; Figure 25.1).

However, the same species of pine had an earlier effect on streamflow (within 3 years) under the drier conditions prevailing in the Mokobulaan B catchment at Mpumalanga. Other conditions remaining the same, eucalypts have a slightly earlier impact on streamflows than pines, normally within 2–3 years (Table 25.3, column 5; Figure 25.1). Like in the case of deforesting humid tropical (Bruijnzeel, 1996) and temperate (Hewlett and Bosch, 1982) catchments, the presently found changes in streamflow following the establishment of exotic timber species into native vegetation are related to the proportion of the catchment affected, the contrast in biomass, rainfall depth (water availability) and growth rate (vigour) of the forest (Smith and Scott, 1992; cf. Sikka *et al.*, 2003).

The high values of plantation water use (E_T) inferred from the afforested South African catchment experiments are confirmed by process studies that have measured transpiration totals of 1300 mm yr⁻¹ and total E_T of 1500 mm yr⁻¹ (Dye, 1996; Burger *et al.*, 1999). Such values are comparable to some of the higher estimates for tropical plantations listed in

Table 25.3. Summary of annual flow reductions in South African afforestation experiments, showing the age from which reductions were first statistically significant (column 5), the peak reductions over a 5-year interval (columns 6 and 7), and the decline in flow reductions towards the end of the rotation or period of measurement (column 8)

1. Catchment and native vegetation	2. Area and trees planted and plant date (all at 1360 sph)	3. Latitude and mid-elevation (m)	4. Median annual rainfall (mm)	5. Initiation of reductions (age, yrs)	6. Peak reductions, absolute (5-yr mean) (mm/10%) (age, yrs)	7. Peak reductions, relative (5-yr mean) (%/10%) (age, yrs)	8. Declining effect in last 5 years ^a (mm/10%; %/10%)
Cathedral Peak II: montane	75%; <i>Pinus patula</i> 1951	29° 0' S 2150	1399	7	67.1 19	7.9 26	2 0.3
Cathedral Peak III: montane grassland	86% <i>Pinus patula</i> 1959	29° 0' S 2100	1399	8	51.7 19	6.6 23	4.1 0.8
Mokobulaan A: seasonal grassland	97% <i>Eucalyptus grandis</i> 1969	25° 17' S 1360	1189	2	41 7	10 14–18	0 0
Mokobulaan B: seasonal grassland	95% <i>Pinus patula</i> 1971	25° 17' S 1360	1197	3	17.3 18	10 15–20+	0 0
Westfalia D: evergreen afromontane forest	98% <i>Eucalyptus grandis</i> 1983	23° 44' S 1280	1182	3	37.3 5	9.8 10	13.6 3.4

^a Declining effect is calculated as the streamflow reduction over the last 5 years of record divided by the maximum streamflow reduction recorded over a 5-year interval during the rotation.

Source: (Modified from Scott *et al.*, 1999).

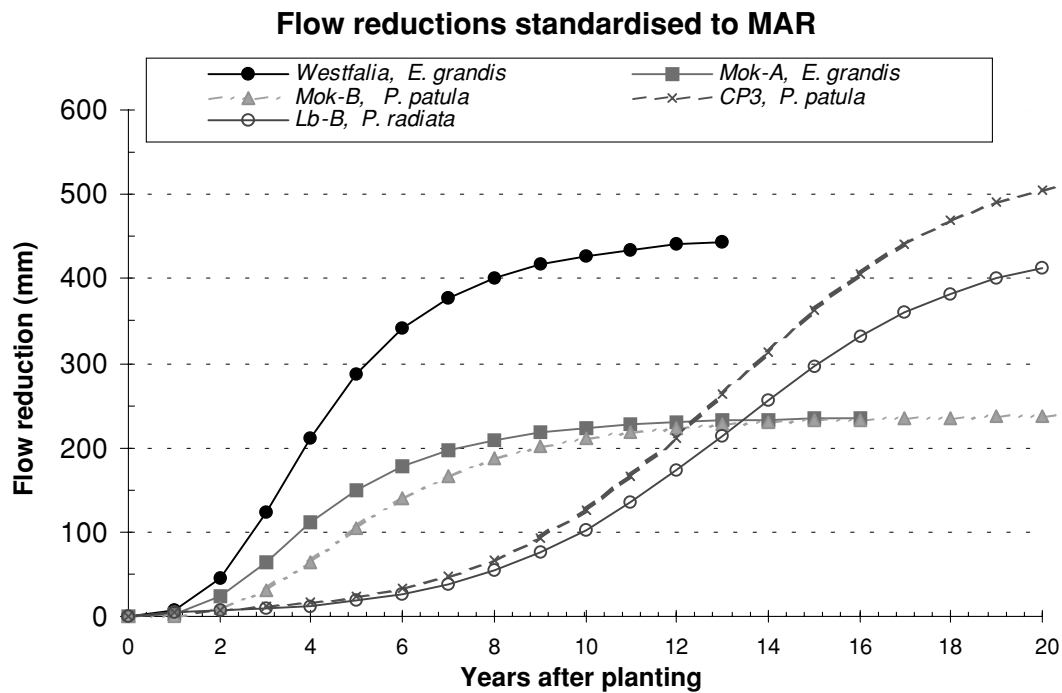


Figure 25.1 Smoothed streamflow reduction curves measured in five South African afforestation experiments where pines and eucalypts replaced grassland, scrub (Lb-B) or native forest (Westfalia). These

curves were generated by plotting the product of measured percentage flow reduction (Scott and Smith, 1997) and mean annual runoff (MAR) of the experimental catchment.

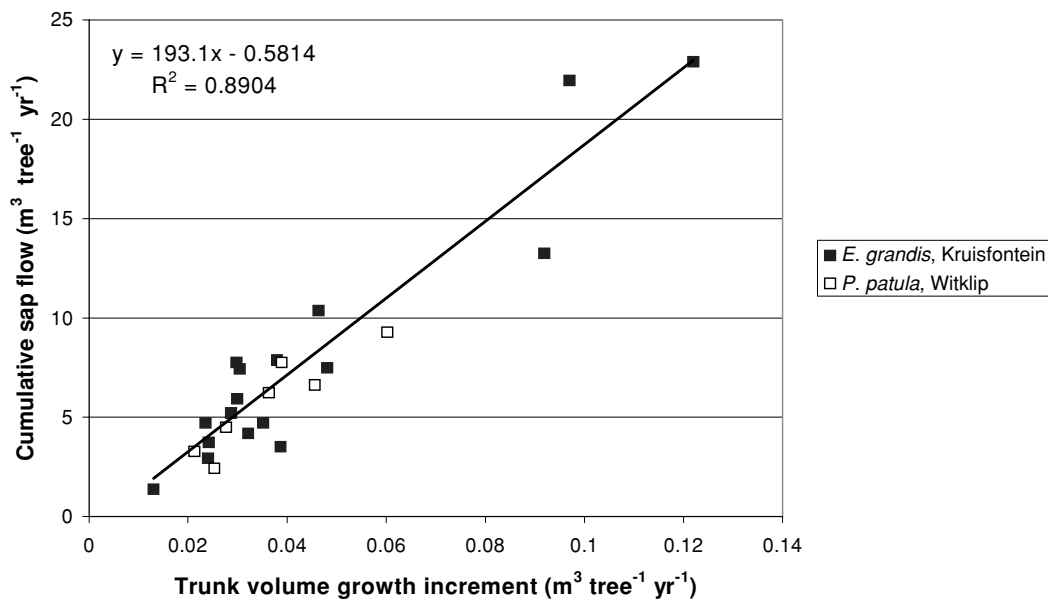


Figure 25.2 Experimental results showing the relationship between trunk volume growth increment and annual cumulative sap flow

Table 25.2. Therefore, it is feared that in sub-humid catchments (MAP < 1200 mm) in the region all streamflow will cease if entire catchments are planted, particularly during the dry season (cf. Figure 25.5 below).

The link between productivity and water use

The generalised curves of Scott and Smith (1997) describing the changes in streamflow reduction over time (Figure 25.1) in the South African experiments closely resemble growth curves for these species (Scott and Lesch, 1997; Dye *et al.*, 2001). The rapid desiccation of the catchments is attributed more to the increased transpiration of the tree crop than to an increase in interception losses (Dye, 1996; Scott and Lesch, 1997). The exotic trees are evergreen and have deeper root systems, which allow them to sustain transpiration year-round, while native vegetation would have closed its stomata in response to dry season stress (Dye, 1996; cf. Waterloo, 1994). Recent work by Dye *et al.* (2001) has shown the strong link between the growth rate of these plantation trees and their transpiration (Figure 25.2). Their findings also support a contention raised in the early review by Bosch and Hewlett (1982) that the magnitude of the responses in water yield to forest clearing or establishment was positively related to the productivity of the forest. Other examples of the close connection between stand productivity (using leaf area index as a proxy) and water use include the studies of Dunin *et al.* (1985) on eucalypts in south-east Australia, Granier *et al.* (1992) on young plantations of two indigenous species in French Guyana, and Waterloo *et al.* (1999) on grassland and *Pinus caribaea* in Fiji.

recorded in sample trees of *Eucalyptus grandis* and *Pinus patula* in the Mpumalanga Province of South Africa. (Dye *et al.*, 2001).

A new finding from the up-dated analysis of the South African afforestation experiments (Scott *et al.*, 1999, 2000), is that flow reductions are definitely diminished towards the end of longer timber rotations, and this applies both to pines and at least one eucalypt experiment (Table 25.3, column 8). Obviously this trend is clearest in the longer-term experiments. The diminution of final flow reductions (mean values over the last 5 years measured) compared to the highest 5 year average reduction at any one time during the rotation ranges from zero (no change over time, usually in short-term experiments) to 60% and 50% less, for absolute and relative measures, respectively (Table 25.3).

The finding of streamflow increasing again with plantation age during the post-maturation phase of the trees agrees with the Australian experience in regenerating mountain ash (*Eucalyptus regnans*) forests near Melbourne, Victoria, following the 1939 wild fires (Langford, 1976; Kuczera, 1987; Jayasuriya *et al.*, 1993). Water yields from the old-growth (>150 years) forest were nearly 60 mm yr⁻¹ per 10% forest cover higher than those from the young, vigorously regrowing forest that developed after fire (Figure 25.3). Whilst the mountain ash forest results are in apparent contradiction to the generalisation that streamflows decrease in proportion to increasing forest cover, the observed water yield declines in the first 25–30 years during the regrowth of the mountain ash forest in fact parallel the flow reductions observed after the establishment of plantations in South Africa (cf. Figures 25.1 and 25.3). However, the accent in the Australian work has always been on the (very) mature forest stages, which are accompanied by flow increases with forest age that justify the maintenance of a

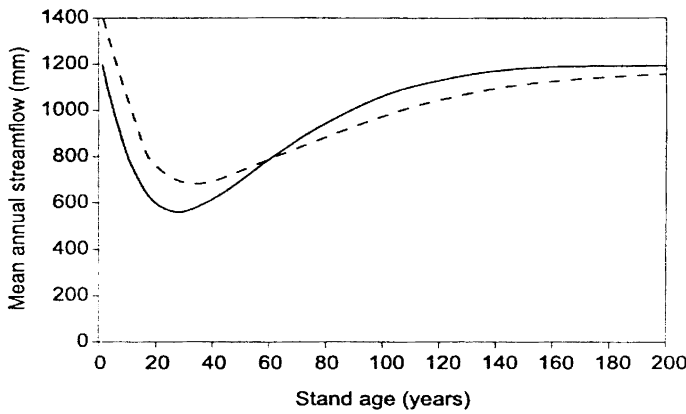


Figure 25.3 Two models of the response pattern of streamflow over time after fire in mountain ash (*Eucalyptus regnans*) forest north of Melbourne, Australia. (After Watson *et al.*, 1999.)

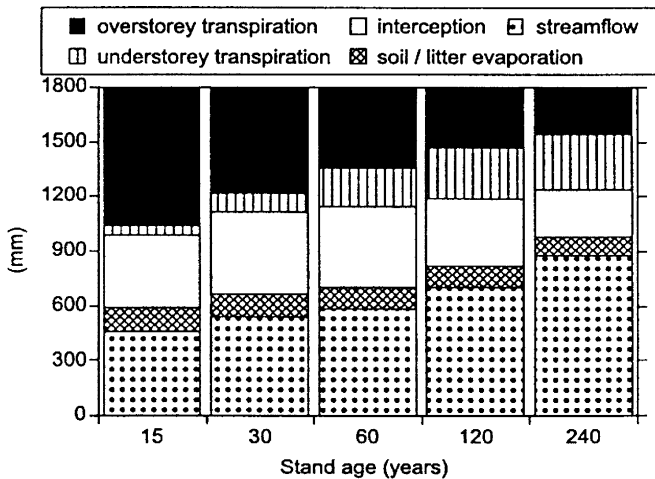


Figure 25.4 Changing contributions of various evaporation components with age in mountain ash forest, Melbourne, Victoria, Australia. (After Vertessy *et al.*, 1998a.)

forest cover in these catchments whose main purpose is the supply of water to greater Melbourne.

Process studies by Haydon *et al.* (1996) and Dye (1996) show that increases in eucalypt age are associated with declines in leaf area index, sapwood conducting area and transpiration per unit of leaf area. Modelling of the various evaporation components in the mountain ash forests over time further shows that transpiration by the overstorey dominates the water balance during the initial 15–30 years, but declines gradually over time to end up being less than interception losses in over-mature forest (Vertessy *et al.*, 1998a, b) (Figure 25.4).

The implication of this is that in highly productive or vigorously growing forests, transpiration can be expected to dominate the water balance, total water use will be at its highest, and streamflow reductions at a maximum. As the forest matures, so transpiration

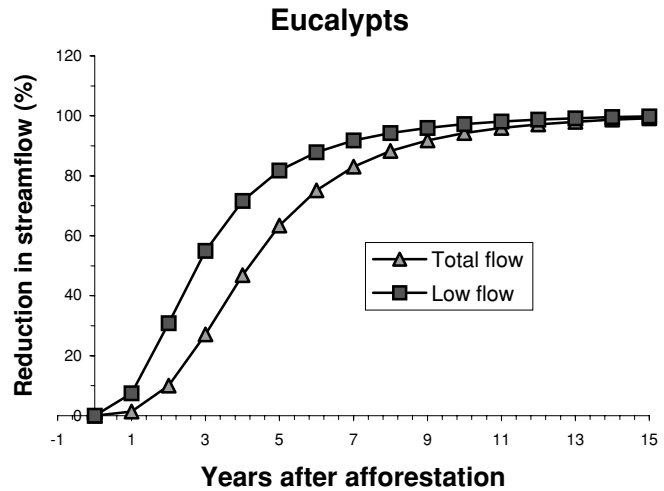


Figure 25.5 Smoothed curves fitted to observed streamflow reductions measured in two experimental catchments planted to *Eucalyptus grandis* in South Africa, showing the larger and earlier effects of afforestation on the low flow component. (After Scott and Smith, 1997.)

will decline, though it might be compensated for by increases in interception losses and understorey transpiration (Figure 25.4). Overall, though, mature forest of lower vigour can be expected to have higher streamflow yields than when it was younger.

Two other well-known phenomena are also accounted for by this hypothesis. In Australia, pine plantations are known to have a larger impact on streamflows than the native eucalypts in a similar climate (Vertessy, Zhang and Dawes, 2003). This, it has been suggested, is because the productive pine plantations are being compared to the natural (old-growth) eucalypt forests, which may have a higher biomass and greater structural complexity (and hence a higher interception loss) but are less vigorous (productive) and thus transpire less (cf. Vertessy *et al.*, 1998a, b). However, the productivity of the native eucalypt forests of Australia cannot be compared to that of the disease-free, mono-specific eucalypt plantations of South Africa, Brazil or other tropical and subtropical sites, where growth rates in excess of 30 m³ ha⁻¹ yr⁻¹ have been recorded (Lima, 1993; Dye, 1996). Interestingly, a similar change in water use with age is now beginning to be revealed for regenerating tropical rainforest vegetation as well. Here too, there are indications that water use by very young and vigorously growing secondary vegetation is similar to, or even higher than that of old-growth forest in the same area (see discussion in Hölischer *et al.*, this volume; Giambelluca, 2002).

Effect of forestation on low flows

As pointed out earlier, the bulk of the streamflow increases resulting from experimental forest clearing are manifested as baseflows rather than in the form of greatly increased stormflows. Figure 25.5

illustrates the similarity in the pattern of reductions in total and dry season (low) flows found for two catchments afforested with eucalypts in South Africa, though low flows are decreased more than are total flows at the same age. The effect of forestation on low flows has two supposed sources. First, these and other exotic plantations, in contrast to the native vegetation they replace, do not go dormant in the dry season (Smith and Scott, 1992; cf. Waterloo *et al.*, 1999). The second component, though less easily quantified, is that of steadily reducing soil water stores through several years, as found under eucalypts in India (Calder *et al.*, 1997; Sharda *et al.*, 1998) and in South Africa (Scott and Lesch, 1997). Low flows are a reflection of the amounts of soil water and groundwater stored in the catchment and as these are steadily depleted by tree water uptake so will low flows diminish accordingly. It is clear from the South African experiments that total water use by the tree crop can exceed annual rainfall in many years (Dye, 1996; Burger *et al.*, 1999; Dye *et al.*, 2001), and that, once summer streamflow has ceased altogether (Figure 25.5), the occurrence of rainstorms may not easily cause the streams to flow again. Once again, the changes in flows observed in South Africa reflect differences in vegetation water use only, not in soil infiltration capacity. The possibility of improved groundwater recharge through enhanced infiltration afforded by tree planting on degraded soils will be discussed below.

The greater effect of forestation on low flows observed in South Africa has also been reported for plantations of *Pinus caribaea* in seasonal grasslands in Fiji (Kammer and Raj, 1979) as well as under *Eucalyptus globulus* in South India (Sharda *et al.*, 1988), particularly after coppicing (Sharda *et al.*, 1998; Sikka *et al.*, 2003). It was also recorded after afforestation of an extensive upland area in northern Malawi where between 1966 and 1976 almost 93% of a 13.3 km² grassland catchment was progressively planted with a mixture of *Pinus patula*, *P. kesiya* and *Eucalyptus saligna*. Streamflow records for nine pre-planting years and 13 years following planting, showed no significant change in peak discharges, but the 10 and 30-day low flow totals showed a significant reduction over this period (Mwendera, 1994). Thus, the finding of strongly reduced baseflows after forestation of non-degraded grasslands can be expected to be generally applicable and should be heeded when planning industrial plantations (Calder, 1999).

Effects of forestation on storm flows

Forest hydrological research has shown that the influence of vegetation cover or type on stormflows is inversely related to the size of the rainfall event that generates the flows. Hewlett and Doss (1984) showed this to be the case for a broad range of humid catchments of the eastern USA, whereas Hewlett and Bosch (1984) demonstrated a similar independence of stormflows on vegetation cover

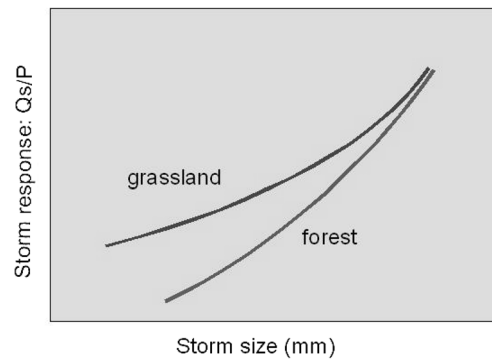


Figure 25.6 Postulated relationship between size of a stormflow generating event (P) and the resultant stormflow (Qs) and how these are affected by vegetation type under non-degraded topsoil conditions.

in afforested and control catchments in South Africa. Likewise, peak flows in the lower ranges were affected by the establishment of a vigorously growing eucalypt plantation in a montane grassland catchment in south India but the effect was almost negligible in the higher ranges (Sharda *et al.*, 1988; Sikka *et al.*, 2003). Soil infiltration capacities were not affected appreciably by the land-cover change in any of these studies, however, nor were groundwater levels in the swampy valley bottoms that produced most of the storm runoff in the Indian study (Samraj *et al.*, 1988).

In small storm events the combined storage capacity of vegetation canopies, ground-covering litter, surface micro-topography and the soil mantle can be substantial in proportion to the size of the storm depth. Of these the soil mantle is potentially the largest water store, but its capacity to accommodate additional rain varies as a function of soil wetness. Where previous uptake by the trees has depleted soil water reserves, storage capacities will be relatively high but once the soil has become thoroughly wetted by frequent rains (typically at the height of the wet season), opportunities to absorb large additional amounts of rain will be limited even under fully forested conditions. Furthermore, as precipitation events increase in size, so does the relatively fixed maximum storage capacity of the soil become less influential in determining the size of the stormflows that are generated (Figure 25.6). In other words, under conditions of extreme rainfall and soil wetness, large stormflows may also emerge from forested areas (Hewlett, 1982; Hamilton and King, 1983; cf. Bonell, this volume).

It can be seen, therefore, that the influence of vegetation on the magnitude of stormflows will be determined by its proportional contribution to the total storage capacity of the catchment, but in large storm events this contribution may easily become unimportant (Hewlett, 1982; Bruijnzeel and Bremmer, 1989). However, as discussed more fully in the next section, where degradation of a catchment has produced strong reductions in canopy and ground cover (including litter), and above all in infiltration capacity and soil depth through continued erosion (and thus overall soil

water storage opportunity), reforestation could clearly lead to an improvement of most or all these factors over time.

FORESTATION OF DEGRADED LANDS: PROSPECTS FOR IMPROVED FLOW REGIME

The term 'degraded' refers to areas where the imposed land use has caused vegetation to be strongly simplified in terms of diversity and structure and where such changes are associated with soil compaction or crusting and erosion, such that surface hydrological processes are negatively affected (Lal, 1987). The expected hydrological changes would include a more or less pronounced reduction in infiltration capacity and thus increased contributions by infiltration-excess overland flow to stormflows, usually an increase in total water yield but with a decreased baseflow component compared to the pre-degradation situation because of the deteriorated infiltration and soil water recharge opportunities (Pereira, 1989; Sandström, 1998). Physical soil degradation is a widespread phenomenon, particularly in the dry tropics (Eswaran, Lal and Reich, 2001), but also under humid tropical conditions. For example, about 45% of the total land area of South and South East Asia was considered to be affected by some form of human-induced soil degradation (mostly surface erosion) in the mid 1990s. On 10–15% of the affected area, degradation was considered to have a strong to very strong impact on plant productivity (and by implication, on overland flow occurrence), whereas on 22–28% the impact was considered moderate. Impacts were light on 40–48% of the land and considered negligible on the remaining 12–23% of the land (Van Lynden and Oldeman, 1997).

Reforestation is often recommended as a means of reducing the enhanced surface water losses associated with soil degradation, returning water to a subsurface route through the soil profile, thereby ultimately restoring baseflows (e.g. Bartarya, 1989; Negi, Joshi and Kumar, 1998). The question is whether this process really occurs, in which circumstances might it occur, and would it also occur if forestation is by means of a productive timber plantation of high water uptake? The answer to this question is that baseflow is only likely to be improved if gains through enhanced infiltration after forestation exceed the associated increase in evaporative losses (Bruijnzeel, 1989b). The information on excess plantation water use over that of the grassland or rain-fed crops they replace (Tables 25.2 and 25.3) suggests that infiltration volumes would need to be increased by at least 175–200 mm yr⁻¹ under subhumid tropical conditions (MAP 1200–1400 mm) and by as much as 450–700 mm yr⁻¹ under wetter conditions (MAP > 2000 mm). Whether this can be attained depends on the size of the overland flow component, which, in turn, is largely decided by the prevailing climate (notably rainfall intensity and duration),

the degree of soil degradation, and the nature (texture, erodibility, depth) of the soils. Together with the ability of the new vegetation to create additional soil water storage opportunity through enhanced water uptake, these climatic and soil factors will determine the extent to which stormflow components (overland flow, subsurface stormflow) can be reduced by a change in vegetation cover (Bailly *et al.*, 1974; Chandler and Walter, 1998; cf. Bonell, this volume). It is important to note that it is not necessarily the vegetation cover per se that will be responsible for a change in near-surface hydrological behaviour, but rather the changes (or not) in associated soil characteristics such as topsoil bulk density and infiltrability, soil depth, as well as surface detention and retention storage. The following three examples from East Africa illustrate this.

Working under semi-arid (MAP c. 800 mm) tropical conditions in northern Tanzania, Sandström (1998) concluded (mostly on the basis of model simulations) that where soils were fine-textured and liable to crusting upon exposure to rainfall, the creation and maintenance of macropores by trees within a setting of closed woodlands was crucial for effective groundwater recharge. By contrast, in a degraded nearby catchment deforested more than 40 years ago, overland flow formed a large component of the water balance, and overall groundwater recharge was strongly diminished despite the fact that water use by the non-forest vegetation must be much reduced. This led to perceived desiccation of the deforested catchment although the soils were deep and potentially had a high capacity to store water. Thus, surface characteristics rather than the contrast in vegetation water use proved to be the dominant factor here. A notable aspect of this study is the long time between the clearing of forest and the measurements. It is likely that the degradation of the soil hydraulic properties of the cleared catchment represents a gradual process that might not have been apparent closer to the time of the original conversion (Sandström, 1998). Indeed, at a somewhat wetter (MAP 1960 mm) site elsewhere in Tanzania, Edwards (1979) did not observe significant surface soil degradation nor diminished annual or low flows throughout an 11-year period of subsistence cropping. In fact, both total and summer flows were strongly increased after clearing (Table 25.2). He attributed this to a fortuitous combination of the low rainfall intensities prevailing at this montane location and the low erodibility of the deep volcanic soils. Similarly, in a broader scale study of rainfall partitioning and groundwater recharge in nearby central Uganda, Taylor and Howard (1996) estimated that of an annual rainfall of 1400 mm, around 200 mm became recharge by infiltration of rain water. Moreover, using a combination of isotope tracing and modelling, the authors were able to determine that recharge had doubled over a period of 30 years as a result of 26% of the catchment area having been converted from forest to mixed agricultural use. In this and the previous instance, the surface properties of the affected land at the catchment scale

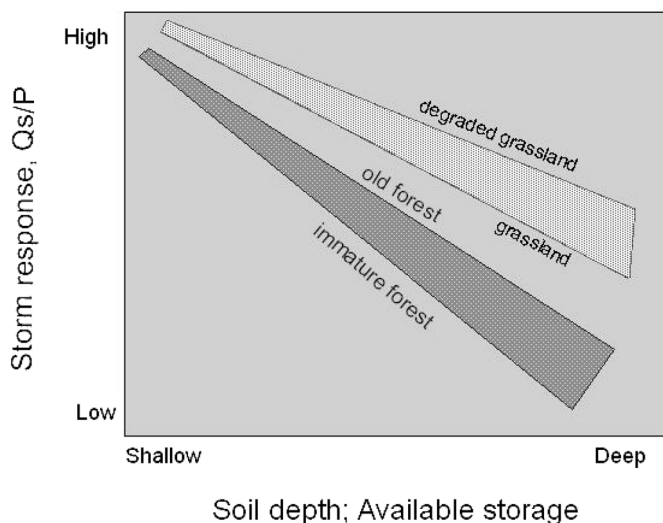


Figure 25.7 A postulated relationship between catchment storage capacity and stormflow response, and the likely role of vegetation cover.

apparently remained suitable for water to move deeply through the soil profile rather than over the surface, thereby allowing the reduction in vegetation water use to become manifest as increased groundwater recharge and baseflows. Interestingly, the magnitude of the changes in recharge or flow at the two locations was very similar at 40–45 mm yr⁻¹ per 10% forest removal (Taylor and Howard, 1996; Edwards, 1979).

The degree of surface runoff reduction required to affect dry season flows under humid tropical conditions (MAP 2500 mm) was examined through modelling by Van der Weert (1994). He simulated the relative contributions to annual water yield by three streamflow components, viz. fast surface runoff (overland flow), delayed subsurface flows (called ‘interflow’ by Van der Weert), and deep groundwater outflow (baseflow) for a large river basin in West Java, Indonesia, under fully forested and cleared conditions as well as for gradually increased surface runoff coefficients. The simulations showed that baseflow levels would be little affected by land use change as long as the overland flow coefficient remained below 15% of the rainfall (equivalent to *c.* 350 mm yr⁻¹). However, should surface runoff become as high as 40% (close to 1000 mm yr⁻¹), baseflow (dry season flow) would be roughly halved. These estimates are almost double the amounts inferred from contrasts in vegetation water use only (Tables 25.2 and 25.3). A second modelling exercise by the same author suggested that dry season flows would diminish more rapidly following severe surface disturbance of deep soils with a large storage capacity than in the case of more shallow soils having little capacity to store water anyway (Van der Weert, 1994; cf. Sandström, 1998). Figure 25.7 is a conceptual attempt to represent the combined effect of soil depth (storage) and land cover (infiltration) on catchment stormflow response (cf. Table 25.4).

There is very little direct information on the hydrological behaviour of degraded catchments in the humid tropics, at least in the ‘official’ literature. Some idea may be gained, however, from the stormflow coefficients reported in the grey literature for ‘typical’ actively eroding (‘semi-degraded’) agricultural catchments in the volcanic uplands of Java, viz. 30–40% of incident rainfall vs. 5–10% for nearby forested catchments (Pramono Hadi, 1989; Rijdsdijk and Bruijnzeel, 1991; Jongewaard and Overmars, 1994). Likewise, at *c.* 35–50 t ha⁻¹ yr⁻¹ the sediment yields from such deforested catchments are 10–50 times higher than those typically associated with similarly sized forested catchments (cf. Figure 22.6 in Grip *et al.*, this volume). Such increases largely reflect the enhanced contributions of HOF from degraded agricultural fields and compacted surfaces like trails, roads and settlements which all exhibit very low infiltration capacities (Grip *et al.*, this volume). However, whilst storm runoff in the most extreme cases can be increased to as much as 70% of the rainfall (e.g. where the underlying rock has become exposed), the associated sediment yields are relatively low because most of the erodible material has already been lost (Flatfjord, 1976). Such examples illustrate the very considerable (local) potential for highly increased storm runoff from severely degraded catchments.

There are even fewer studies of how, and to what extent, a change in land use may improve low flows (Bruijnzeel, 2004). Therefore, the remainder of this section explores to what extent infiltration and soil water recharge may be boosted (or stormflows reduced) after forestation of degraded land. These amounts may then be compared with the expected increases in water use (Tables 25.2 and 25.3) in a first attempt to derive when the net result will be positive (i.e. increased low flows) or negative (i.e. further decrease in low flows).

Table 25.4 summarises the available evidence with respect to the reductions in stormflow volumes observed after forestation or soil conservation works in a series of degraded (sub)tropical catchments that were mostly too small to sustain perennial flow. As such, the corresponding effects on low flows cannot be evaluated directly. The evidence collated in Table 25.4 does show that major relative (and sometimes also absolute) reductions in stormflows may be achieved by forestation and various soil conservation measures (and *vice versa* through adverse practices like overgrazing or repeated burning). However, almost none of these reductions seem to be large enough in absolute terms to overcome the typical increases in vegetation water use after forestation inferred earlier (175–200 and 450–700 mm yr⁻¹ in subhumid and humid areas, respectively (Table 25.2)), let alone the values implied by Van der Weert (1994) in his simulations (up to 1000 mm yr⁻¹). It is probably pertinent that the only exception (Leyte, Philippines; see below) represents a high rainfall site (MAP 2200 mm). Usually, as for example in southern China (MAP 1500 mm), ‘direct’ stormflows are reduced after forestation compared to severely degraded

Table 25.4. Effects of forestation or soil conservation works on stormflow volumes (and their components) in selected small degraded (sub)tropical headwater catchments

Location	Vegetation and treatment	Area (ha)	Actual stormflow			Change in total storm flow vs. baseline (%)	Reference and notes
			HOF	SSF	Total		
Leyte, Philippines	Degraded pasture	0.13	1518	154	1672	Baseline	Chandler and Walter (1998): direct comparison of hillslope plots with contrasting cover, shallow soils, MAP c. 2200 mm
	Pasture + hedgerows	0.23	396	286	682	-59	
	Conservation cropping	0.25	242	132	374	-78	
	Conservation cropping (mulched)	0.13	66	220	286	-83	
Luzon, Philippines	Regrowth (10–20 yrs)	0.20	<22	44	<66	-96	Daño (1990): paired catchment, sequential treatment on experimental catchment, firstly 4 years of no-burn, secondly reforestation (8 years), MAP c. 3800 mm, catchments highly leaky (Bruijnzeel, 1990).
	Burned grassland	1.6	-	-	1244/905	Baseline	
	Protected grassland	1.0	-	-	1359	+9	
	6-yr <i>Gmelina</i> forest	1.0	-	-	1002	+11	
Guangdong, China	Badland	3.7	151	589	740	Baseline	Zhou <i>et al.</i> (2001): direct comparisons, total stormflow equated to measured non-perennial flow totals (Q_t) and will include some delayed flow; HOF equated to direct runoff (Q_d) based on traditional hydrograph separation into surface runoff, throughflow and baseflow by Zhou <i>et al.</i> ; $SSF = Q_t - Q_d$; deep soils, MAP c. 1455 mm.
	<i>Eucalyptus</i>	3.8	90	241	330	-55	
	<i>Eucalyptus</i> plus other trees	6.4	<0.5	41	42	-94	
Madagascar	Burned grassland	3.25	-	-	221	Baseline	Bailey <i>et al.</i> (1974): direct comparisons, results probably influenced by differences in leakage, MAP 1717 mm
	Protected grassland	3.18	-	-	121	-45	
	Conservation cropping	4.77	-	-	48	-78	
	<i>Pinus patula</i> (<10 yr)	3.90	-	-	42	-81	
North India	Degraded scrub	1.27	-	-	37/70	Baseline	Gupta <i>et al.</i> (1974, 1975): paired catchments, coefficients of determination for calibration and treatment equations unknown; runoff totals pertain to rainy seasons of 1974/75 with rainfall totals of 623 and 718 mm, respectively.
	Annual burning	4.15	-	-	66/57	+218/26	
	Overgrazing	0.74	-	-	73/117	+552/30	
	Forestation/trenching	1.93	-	-	17/12	-29/-75	

land (in this case by 60 and 150 mm yr⁻¹ after forestation with eucalypts and mixed species, respectively) but both delayed and total (ephemeral) flows were decreased as well (Table 25.4). As such, no improvement in baseflows was obtained. Most probably, this reflects the higher water use of the mixed forest although measurements of soil water (to a depth of 200 cm) and groundwater levels suggested both to be lower under the pure eucalypt plantation (Zhou *et al.*, 2001). The importance of a well-developed litter layer and understorey vegetation for minimising infiltration-excess overland flow (HOF) and surface erosion (cf. Wiersum, 1985) was also demonstrated by the Chinese study. A patch with very little litter and shrub cover was protected from litter harvesting (litter is collected routinely for use as fuel in this region). After two years, litter biomass already attained 90% of its value after eight years whilst understorey cover continued to increase exponentially throughout this time. HOF was reduced from 31% of incident rainfall at the start of the experiment to less than 9% after eight years. The more completely developed forest floor in the mixed plantation exhibited negligible overland flow and surface erosion (Zhou *et al.*, 2001). Thus, this long-term study (>10 years of measurements) demonstrates the effectiveness of using eucalypts as a pioneer crop in restoring at least some hydrological functions in degraded catchments in this part of China, particularly the hydrological benefits of allowing litter accumulation on site and of encouraging succession to a mixed forest ecosystem. Once again, no improvement in low flows was achieved, however.

The finding of Daño (1990) of *increased* runoff after protecting and subsequently reforesting a very small (and very leaky) annually-burned grassland catchment in the Philippines is difficult to explain and is at odds with the previous results as well as those obtained by Bailly *et al.* (1974) and Gupta *et al.* (1974, 1975) for similar (but drier) situations in Madagascar and northern India, respectively (Table 25.4). One would have expected the higher water use of protected grassland and fast-growing trees to result in both improved infiltration and higher water use, and thus into less overall water yield. Such results illustrate the inherent limitation of the paired catchment approach which, although statistically sound, is essentially a black box (Bruijnzeel, 1996). Elsewhere in the Philippines, and at the other end of the range, Chandler and Walter (1998) reported massive reductions in stormflow (HOF plus subsurface stormflow, SSF) from degraded hillslope-sized plots after being subjected to various conservation practices whereas the smallest volumes were observed under 10–20-year-old secondary forest. However, the size of the inferred reductions (which are based on direct comparisons between plots rather than previous intercalibrations) hinges very much on the very high runoff total established for the most degraded plot (1670 mm year⁻¹ or 76% of incident rainfall) (Table 25.4). Also, it is not clear whether the soil of the forested plot had been equally degraded before abandonment. Nevertheless, taking the runoff reductions observed by

Chandler and Walter (1998) at face value, at 990–1605 mm yr⁻¹ they should be more than sufficient to compensate any increases in evaporative losses compared to degraded grassland (presumably <750 mm yr⁻¹) (Table 25.2). It is interesting to note that the large reductions in HOF when going from severely degraded grassland to various types of conservation cropping were not matched by concurrent changes in SSF and delayed throughflow although both HOF and throughflow were lowest under forest (Table 25.4). As such, one may expect a significant portion of the increased infiltration to contribute to the recharging of groundwater reserves (and thus baseflow) rather than to be lost from the soil profile via rapid near-surface flows, although neither groundwater levels nor soil physical characteristics, rainfall intensities or tree water uptake rates were given by Chandler and Walter (1998). Once again, additional process-based work would be needed to obtain more comprehensive answers.

An alternative (non-catchment) approach bases predicted changes in occurrence of overland flow and SSF on a comparison of changes in soil infiltration capacity and saturated hydraulic conductivity (K^*) profiles with depth after forestation vs. rainfall intensity – duration distributions. Although the dataset is still small and covers only a small portion of the pan-tropical spectrum of soil, tree species and rainfall intensity combinations, a few tentative observations seem possible.

Firstly, the recovery of surface K^* following forestation with teak (*Tectona grandis*, usually planted on calcareous heavy clay soils in seasonally dry areas) is generally very limited and HOF (as well as surface erosion) remains rampant (Bell, 1973; Wolterson, 1979; Mapa, 1995). Development of surface organic matter in these forests is often hampered by various forms of disturbance, including grazing, litter collection and, above all, fire (Wiersum, 1984b)². Secondly, where initial values of surface K^* are particularly low (<5–10 mm h⁻¹?), modest increases of c. 3–5 mm h⁻¹ per year since planting or abandonment seem to be typical, regardless whether the trees are planted or regenerate naturally. For example, Giambelluca (2002) reported increases of up to c. 80 mm h⁻¹ over 25 years of forest regeneration after shifting cultivation on red latosols in northern Vietnam whereas under more seasonal climatic conditions and on lateritic soils in Karnataka, India, B. K. Purundara and collaborators found log mean values of K^* to increase from 5.75 mm h⁻¹ under degraded grassland to 22.5 mm h⁻¹ in a five-year-old plantation of *Acacia auriculiformis*, which further increased to 44.0–56.8 mm h⁻¹ after 10–12 years. Since K^* under natural forest in the area is about 250 mm h⁻¹ it may take 50 years for the surface infiltration characteristics

2 It is potentially pertinent that Elsenbeer *et al.* (1999) observed rather high values for surface K^* in a non-disturbed teak plantation of unspecified age (but possibly 8–10 years old; Godsey and Elsenbeer, 2002) under rather wet conditions in Rondonia, Brazil, although the relatively low clay content of the latter soil may also be responsible.

to fully recover (M. Bonell, pers. comm.). Thirdly, under similar climatic but somewhat more favourable initial soil conditions (log mean surface K^* c. 40 mm h^{-1}), much larger increases in K^* (up to 140 mm h^{-1}) have been observed either through direct measurement (Gilmour *et al.*, 1987) or may be inferred from reductions in HOF volumes (Zhou *et al.*, 2001) within 8–12 years after reforestation degraded grass- and scrubland in Nepal and China with pines and eucalypts, respectively. In the Nepalese study, log mean surface K^* under more or less undisturbed forest ranged from 370 to 525 mm h^{-1} . On the basis of the rapid (apparent) increases in K^* attained between 5 and 12 years after tree planting (from c. 51 to 183 mm h^{-1}), it would take as little as 20–30 years to reach pre-disturbance values although the authors suggested that it would ‘undoubtedly be many decades’ (Gilmour *et al.*, 1987). Fourthly, under favourable initial conditions, natural regeneration may induce very rapid increases in surface K^* . Lal (1996a), for example, reported a tenfold increase within five years of regeneration (from 190 mm h^{-1} to 1930 mm h^{-1}) in Nigeria. Lal’s observations would seem to be supported by the much reduced occurrence of HOF under fairly young secondary forest compared to degraded grassland in the Philippines referred to earlier (Chandler and Walter, 1998) (Table 25.4) although Giambelluca (2002) found more modest increases in K^* during forest regeneration in Vietnam. Summarising, a tentative pattern of increased rates of improvement of K^* over time for more favourable initial conditions seems to be discernible from the data although more information is needed to substantiate this and the cited rates of change.

However, the one-dimensional approach outlined above has its limitations for the prediction of changes in hillslope runoff response to rainfall associated with forestation. Naturally, any positive effects of increased surface infiltration capacity on the frequency of occurrence of HOF also depend on prevailing rainfall characteristics. For example, comparing surface K^* values with rainfall intensity-duration distributions, Gilmour *et al.* (1987) concluded that the major increase in surface K^* (by 140 mm h^{-1}) observed 12 years after reforestation a degraded pasture site in the Middle Hills of Nepal with *Pinus roxburghii* would make almost no difference to the frequency of HOF because of the prevailing low rainfall intensities. Furthermore, the method has come under scrutiny of late as it tends to overestimate the actual infiltration opportunities during rainfall (Loague and Kiriakidis, 1997; Yu, 1999). A tropical case in point was provided by Lal (1996a, b) in Nigeria who determined very high values of surface K^* but still observed substantial HOF, presumably because of surface sealing by raindrop impact (Lal, 1987). Additional complications arise from the notoriously high spatial variability of K^* and the inability of saturated permeability values to represent effects on infiltration exerted by spatio-temporal variations in soil water contents related to differences in vegetation water use and exposure to radiation

and rain. A recently developed spatially variable infiltration model (SVIM) has been shown to represent temporal overland flow patterns from hillslope plots and small catchments much better (Yu *et al.*, 1997; Van Dijk, 2002; cf. Yu, this volume). Therefore, a ‘dual-constraint’ approach using on-site values of K^* and rainfall intensities on the one hand, and measured (plot-based) volumes of infiltration, HOF and SSF at the hillslope scale should potentially yield more useful results. The authors are not aware of any such integrated studies within the context of evaluating the hydrological consequences of reforestation degraded land in the tropics although the valiant attempts of Chandler and Walter (1998) and Zhou *et al.* (2001) go some way. Clearly, much more work is needed.

Arguably the most thoroughly documented case of the hydrological effects of reforestation of a degraded catchment is from outside the tropics in the White Hollow Catchment, Tennessee, USA (Tennessee Valley Authority, 1961). Both physical and vegetation restoration works were implemented and streamflows monitored for the next 22 years. However, neither total water yield nor dry season flows increased after catchment rehabilitation, and it was concluded that the additional water use of the recovering vegetation cover balanced improved infiltration (Tennessee Valley Authority, 1961). Restoration of streamflow would only seem to be possible where the water use of the forest does not deviate too much from that of the vegetation it has replaced, and where a large increase in infiltration is effected. In a further two documented cases cited by Bruijnzeel (2004) in Slovenia and the South-eastern USA, reforestation of previously degraded catchments was associated with overall reductions in streamflow, particularly during dry years and during the northern summer. This plus the large increases in water use that have been measured in productive plantations of pines and eucalypts that have replaced grasslands in South Africa, Fiji, New Zealand, India and elsewhere (Tables 25.2 and 25.3), strongly suggest that the likelihood of restoring streamflows in tropical catchments by means of establishing timber plantations, or even some other form of forestation, is most unlikely. *The balance of probability is that the already diminished dry season flows in most degraded catchments are likely to be reduced even further by forestation or the establishment of plantations* (Bruijnzeel, 2004), possibly with the exception of very severely disturbed surface conditions (Sandström, 1998; Chandler and Walter, 1998) (Table 25.4).

In view of the extent of the ‘low-flow’ problem (Bartarya, 1989; Pereira, 1989; Sandström, 1998), the testing of alternative ways of increasing water retention in tropical catchments without the excessive water use normally associated with exotic tree plantations should receive high priority. One could think in this respect of (a combination of) physical conservation measures (e.g. bench terracing with grassed risers, contour trenches, runoff collection wells in settlement areas: Negi *et al.*, 1998; Purwanto, 1999; Van Dijk, 2002; cf. Critchley, this volume), vegetative ‘filter’ strips at

strategic points in the landscape (Dillaha *et al.*, 1989; Van Noordwijk *et al.*, 1998), and the use of indigenous species with potentially lower water use (Negi *et al.*, 1998; cf. Bigelow, 2001), possibly in an agroforestry context in which the trees may also assist in enhancing slope stability (Young, 1989, 1997; O'Loughlin, 1984). Calder (1999) suggested rotational land use, in which periods with forest alternate with periods of agricultural cropping, as a potential way of reducing long-term mining of soil water reserves by the trees. Needless to say, for such an approach to be successful, soil degradation during the cropping phases should be avoided as much as possible. Furthermore, there is the intrinsic problem that for optimum soil improvement large amounts of easily decomposable leaf litter need to be produced. Trees that are capable of doing this (e.g. various leguminous trees such as *Albizia*) are usually also very fast-growing and can be expected to exhibit equally high water uptake rates although information is scarce. Arguably, designing optimal land use strategies that minimise water use and maximise agricultural productivity and streamflow at the catchment scale, constitutes a prime research challenge in the years to come.

FORESTATION EFFECTS ON EROSION AND SEDIMENT YIELDS

Deforestation is typically associated with general disturbance of vegetation, ground cover and soil, leading to an increase in soil erosion and sedimentation (Lal, 1987; Bruijnzeel, 1990; Grip *et al.*, this volume). Generally, the degree of surface disturbance is largely controllable, being determined by local physical conditions and the quality of management (cf. chapters by Thang and Chappell, and Cassells and Bruijnzeel, both this volume). However, the increased streamflow associated with the removal of transpiring and intercepting vegetation cover may also increase sediment yields, particularly from the channel and the riparian zone. Similarly, apart from the reduction in stream transporting power (Globevnik, 1998), forestation should also lead to a gradual reduction in surface erosion and sediment yields to the extent that it improves surface cover, restores soil hydraulic properties and allows for recovery of disturbance scars (including skidder tracks, temporary roads, and shallow landslides). Arguably, the most pronounced effects of forestation may be expected in the case of severely eroding land (as opposed to severely *eroded* land where most of the erodible soil material has been removed already and yields are low again; Flatfjord, 1976). Sediment production under natural, forested conditions may vary widely, however, depending on the relative importance of the respective contributing mechanisms (overland flow, gullying, and mass wasting: Pearce, 1986; Douglas and Guyot, this volume). By the same reasoning, forestation of degraded land can be expected to produce a range of

responses in terms of reducing sediment production, ranging from minor to substantial. Sediment yields for humid tropical forested headwater catchments typically increase in the sequence: granite/metamorphic rocks < young volcanic deposits < marls / claystones (Douglas and Guyot, this volume; Bruijnzeel, 2004; cf. Figure 22.6 in Grip *et al.*, this volume). The highest values are usually associated with tectonically active steep land areas prone to mass wasting or where landsliding combines with widespread overland flow, such as in marly areas under a seasonal rainfall regime. In such geomorphologically active terrain, it is unlikely that forestation will have much of an effect in terms of reducing sediment yields (Bell, 1973; Bruijnzeel and Bremmer, 1989). However, where landscapes are relatively stable the restoration of forest may make a relatively large difference on sediment yields (Tennessee Valley Authority, 1961; Lal, 1987) (Figure 25.8). Therefore, when dealing with the effects of changes in land use on erosion and sedimentation, it is helpful to distinguish between surface erosion, gully erosion, and mass movements, because the ability of a vegetation cover to control these various forms of erosion is rather different.

Surface erosion

This form of erosion is rarely significant in areas where the soil surface is protected against the direct impact of the rain, be it through a litter layer maintained by some sort of vegetation or through the application of a mulching layer in an agricultural context (Wiersum, 1984b). Erosion rates increase somewhat upon removal of the understorey but rise dramatically only when the litter layer is removed or destroyed. The initial effect is rather small due to the effect of residual organic matter on soil aggregate stability and infiltration capacity (Wiersum, 1985) but becomes considerable upon repeated disturbance of the soil by burning, frequent weeding or overgrazing, which all tend to make the soil compacted or crusted, with impaired infiltration and accelerated erosion as a result (Wiersum, 1984b; Zhou *et al.*, 2001). Forestation has been shown to be able to reverse these processes over time, through the restoration of plant cover and with it an accumulation of litter to cover the soil surface and the return of biotic activity (Lal, 1987; Zhou *et al.*, 2001). Although erosion on grasslands in good condition (Fritsch and Sarrailh, 1986) or in agricultural fields with appropriate soil conservation measures on otherwise stable slopes (Paningbatan *et al.*, 1995; Young, 1989; Critchley, this volume) is usually low, Smiet (1987) made the pertinent observation that forests provide greater latitude with respect to protection of the soil surface against erosion as compared to grazing or annual cropping. Whilst the degraded natural and plantation forests of many tropical uplands are still able to fulfil a protective role because gaps are usually rapidly colonised by pioneer species, grazing lands are often prone to fire, overgrazing and landsliding.

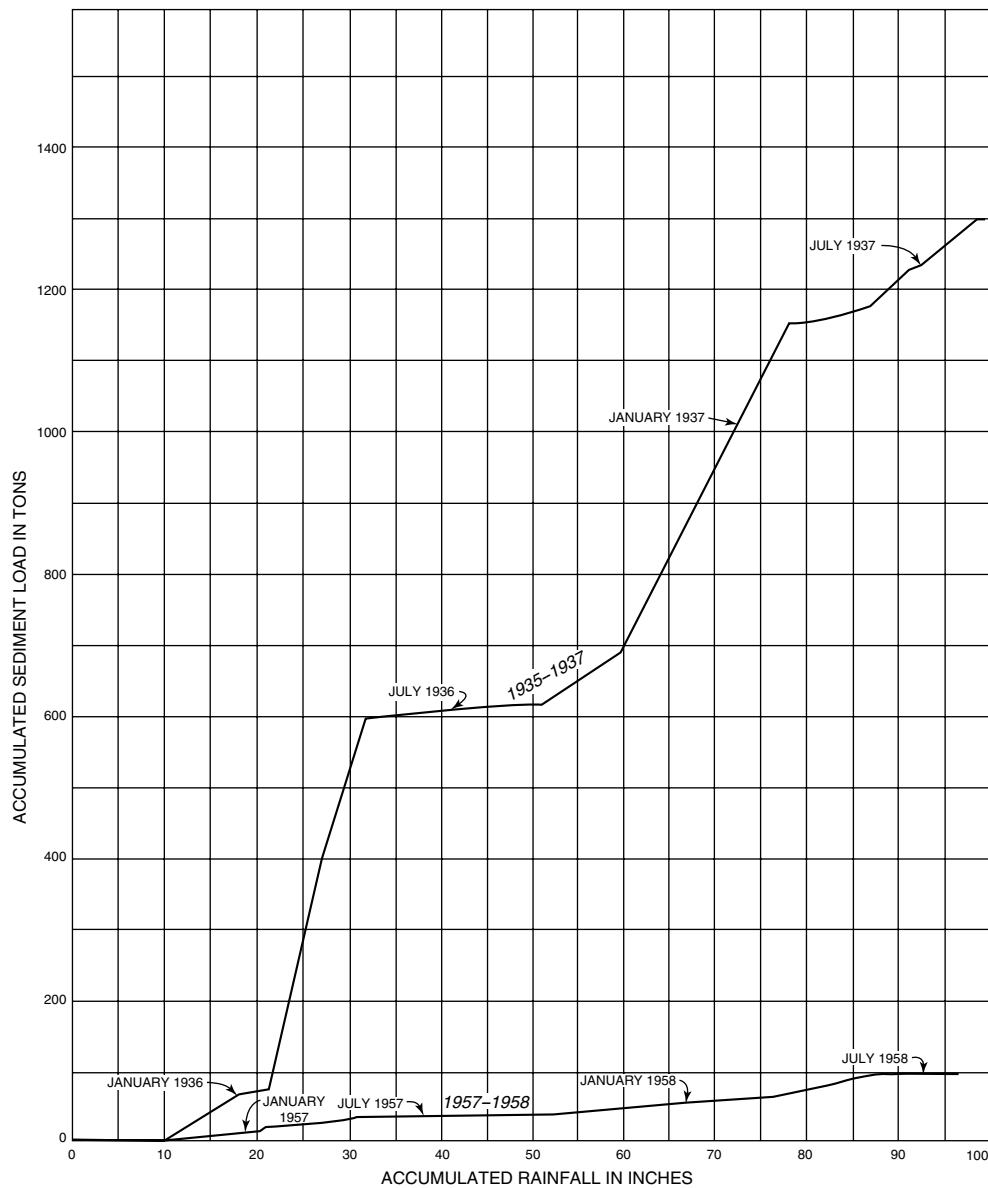


Figure 25.8 Cumulative sediment yield versus cumulative rainfall in the White Hollow catchment, Tennessee, USA before and after

reforestation and other restorative measures. (After Tennessee Valley Authority, 1961.)

There is increasing evidence that erosion rates on and around such compacted surfaces as skidder tracks and log landings, roads, footpaths and settlements can be very high ($35\text{--}500\text{ t ha}^{-1}\text{ yr}^{-1}$; see Bruijnzeel (2004) for details). In addition, the very considerable volumes of runoff generated by such impervious surfaces may promote downslope gully formation and mass wastage. Therefore, contributions of runoff and sediment to the stream network from such areas may be disproportionately large for their relatively small surface area. The lesson for forestation projects is clear, therefore: whilst the overall potential to reduce sediment production is good, frequent disturbance of ground cover, removal of surface litter, and careless harvesting, road-building or road

maintenance, can reverse much of the positive effects of a forest cover. This is particularly important where forestation is part of a conversion to intensively managed timber plantations. During establishment of the new plantation road-building and site preparation, including burning, will tend to increase disturbance and thereby increase the risk of soil erosion and sediment production. Productive plantations can be harvested on a short rotation (as short as 7–10 years for pulp crops in the better eucalypt plantations in South Africa and Brazil; Gonçalves *et al.*, 1997), introducing the potential for regular disturbance associated with harvesting and road works. During the active growth phase of the plantation, there is a strong potential for desiccation of the catchments and

improved ground cover, and as a result, lower overland flow risk and quickflow volumes, and consequently lower sediment yields.

Gully erosion

This is a relatively rare phenomenon in most tropical rainforests but may be triggered during extreme rainfall when the soil becomes exposed through treefall or landslips. In other cases, gullies may form by the collapse of subsurface soil pipes (Douglas and Guyot, this volume). Active gullying in formerly forested areas is often related to compaction of the soil by over-grazing or the improper discharging of runoff from roads, trails and settlements. Alternatively, gullies may develop in the centre of landslides where surface flow tends to concentrate, or when topsoil is removed during forestry operations by heavy machinery leaving more erodible subsoil material exposed (Bruijnzeel, 1997). If gullies are not treated at an early stage, they may reach a point where restoration becomes difficult and expensive. The moderating effect of forestation on actively eroding gullies is limited, however, and additional mechanical measures such as check dams, retaining walls and diversion ditches will be needed (Blaisdell, 1981; FAO, 1985, 1986).

Mass wasting

Mass wasting in the form of deep-seated (>3m) landslides is not influenced appreciably by the presence or absence of a well-developed forest cover. Geological (degree of fracturing, seismicity), topographical (slope steepness and shape) and climatic factors (notably rainfall) are the dominant controls (Ramsay, 1987a, b). However, the presence of a forest cover is generally considered important in the prevention of shallow (<1m) slides, the chief factor being mechanical reinforcement of the soil by the tree root network (O'Loughlin, 1984). Bruijnzeel and Bremmer (1989) cite unpublished observations by I. R. Manandhar and N. R. Khanal on the occurrence of shallow landslides in an area underlain by limestones and phyllites the Middle Hills of Nepal. Most of the 650 slips that were recorded between 1972 and 1986 had been triggered on steep (>33°) deforested slopes during a single cloud-burst whereas only a few landslides had occurred in the thickly vegetated headwater area. However, under certain extreme conditions, such as the passage of a hurricane, the presence of a tall tree cover may become a liability in that trees at exposed locations may be particularly prone to becoming uprooted, whereas, in addition, the weight of the trees may become a decisive factor once the soils reach saturation. Scatena and Larsen (1991) reported that out of 285 landslides associated with the passage of Hurricane Hugo over eastern Puerto Rico, 77% occurred on forest-covered slopes and ridges. More than half of these mostly shallow landslips were on concave slopes that had received at least 200 mm of rain. Brunson *et al.* (1981) described a similar case in eastern Nepal where mass wasting on steep forested slopes was much more intensive than in more gently sloping cultivated areas. Although often occurring

in large numbers, such small and shallow slope failures usually become quickly revegetated and, because of their predominant occurrence on the higher and central portions of the slopes, contribute relatively little to overall stream sediment loads, in contrast to their more deep-seated counterparts (Ramsay, 1987a).

Summarising, the effect of forestation on catchment sediment yield depends on a large number of factors, most notably the relative contributions by surface erosion, gullying and mass wasting. Whilst degraded areas with widespread gullying or massive landsliding will continue to produce significant amounts of sediment following forestation, effects can be expected to be rapid and beneficial in the more typical case of rampant surface erosion and occasional gullying. The restoration of the White Hollow catchment in Tennessee, USA, provides a case in point. Despite serious surface erosion and (some) gullying and bank erosion, sediment yields from this degraded semi-forested and overgrazed catchment responded to reforestation and gully stabilisation works within two years and stabilised at a very low level ever since (Figure 25.8).

CHANGES IN SOIL CHEMICAL CHARACTERISTICS WITH LAND COVER CHANGE

The establishment of tropical tree plantations, like any other form of land use replacing old-growth forest or (degraded) grassland, leads to major changes in nutrient fluxes within and out of the ecosystem, and therefore in soil nutrient reserves (Fölster and Khanna, 1997). Whether these changes degrade or improve soil fertility depends on the original land use, the new land cover, the management practices applied as well as the type of soil and its inherent fertility level. As discussed by Bruijnzeel (1991) and Proctor (this volume), nutrient losses via drainage from undisturbed old-growth rainforests depend on the fertility of the substrate, with larger losses being associated with more fertile substrates. Site fertility is potentially threatened upon forest conversion by the bulk removal of nutrients in harvested timber, through surface erosion due to soil disturbance, and via enhanced leaching. The latter is caused by a combination of (i) the larger volumes of water percolating through the soil after opening up of the canopy (due to reductions in rainfall interception and water uptake), (ii) a temporarily reduced capacity of damaged or sparse vegetation cover to take up nutrients, and (iii) the sudden addition of large amounts of fresh organic debris to the forest floor that are either left to decompose or be burned to facilitate future access for planting (Bruijnzeel, 1998). As such, the establishment of a tree plantation on rainforest land can be expected to set in motion a series of processes that will result ultimately in overall decreased soil fertility (Ruhayat, 1989; Spangenberg *et al.*, 1996; Mackensen *et al.*, 2003; cf. Grip *et al.*, this volume).

Moreover, plantations are usually established to harvest wood for timber or pulp and repeated harvesting of the trees inevitably diminishes soil nutrient reserves through the export of nutrients with biomass removal. The associated losses depend on a number of factors, including: site management history, tree species, rate of tree growth, timber nutrient concentrations, rotation length, harvesting method, as well as nutrient additions from atmospheric sources and weathering (Fölster and Khanna, 1997; Mackensen *et al.*, 2003). Naturally, the need to replace nutrients lost in this way assumes increased importance as more fast-growing species are being planted on low-fertility tropical soils (Brown *et al.*, 1997; Mackensen, 1998). On the other hand, reforestation of degraded grasslands by appropriate management practices may result in a general improvement of soil fertility. One mechanism through which this may be attained is by deep roots taking up nutrients from the subsoil that were previously out of reach but now added to the surface soil via leaf fall and subsequent decomposition. However, direct information on soil improvement through tree planting under humid tropical conditions is exceedingly scarce (Young, 1997).

Fölster and Khanna (1997) and Bruijnzeel (1990, 1998) have drawn attention to the methodological difficulties encountered when trying to evaluate soil fertility response to land use change. Basically, two approaches have been followed, viz. (i) monitoring the changes in (top)soil nutrient concentrations over time in one and the same area (e.g. Amir *et al.*, 1990; Gillman *et al.*, 1985) and (ii) the use of so-called 'false time series' in which soil data from a series of stands of different ages are compared (Hase and Fölster, 1983; Buschbacher, 1984; Bruijnzeel and Wiersum, 1985). Each method has its specific difficulties and limitations. The direct sampling approach not only suffers from problems related to high spatial and temporal variability in soil nutrient concentrations (cf. Proctor, this volume) but also tends to become impractical, particularly in the case of longer-term rotations. Chronosequence studies avoid the latter problem but must assume that all sites under consideration were initially comparable if the results are to be meaningful. Needless to say, the high spatial variability that is typically associated with forest soils easily confounds the results obtained with false time series (Hase and Fölster, 1983; Bruijnzeel, 1990; Waterloo, 1994). Even if based on proper sampling, detailed soil chemical information for a single site may give a limited or even erroneous idea of the magnitude of nutrient reserves or the rates of the various processes acting upon them for the forest as a whole (Van Dam, 2001). Fölster and Khanna (1997) and Bruijnzeel (1998) therefore advocated a nutrient budget approach, which compares the balance between nutrient input and output fluxes over a rotation period with adequately characterised soil nutrient reserves. No single study has measured all the respective inputs and outputs over a full rotation period but work conducted on *Pinus caribaea* on former grassland soils in Fiji (Waterloo, 1994) and *Acacia mangium* replacing logged-over

rainforest in Sabah (Malmer and Grip, 1994; Malmer, 1996; Nykvist *et al.*, 1994) come close. Mackensen *et al.* (2003) applied the budget approach to evaluate net nutrient losses from stands of *A. mangium*, *Eucalyptus deglupta* and *Paraserianthes falcataria* grown in eight-year rotations in East Kalimantan, Indonesia. Estimates of losses associated with leaching, burning and erosion were based on literature data, whereas elemental concentrations in harvested material and the soils were measured.

This section discusses the changes in soil chemical characteristics occurring during plantation establishment and development in a general manner. Most published studies have focused on the changes associated with the conversion from natural forest to plantation forest, rather than the establishment of forest on degraded tropical grasslands (Waterloo, 1994). The soils of the latter are often so impoverished, however, that fertilisation is required to assure establishment of the trees (Otsamo, 1998). In the following the relevant processes affecting soil nutrient levels during land clearing and plantation establishment, maturation and harvesting are briefly described.

Processes affecting soil nutrient levels during land clearing and plantation establishment

Enhanced mineralisation

Clear-felled vegetation (logging debris) is decomposed on the site and leaching of certain nutrients (notably potassium, sodium and phosphorus) from the decomposing material has been shown to be significant (Ewel *et al.*, 1981; Mackensen *et al.*, 1996; Figure 25.9). Mineralisation rates of leaf litter and fine roots are usually enhanced upon forest cutting because of higher nutrient availability (Malmer and Grip, 1994; Silver *et al.*, 1996), increased soil temperatures (Lal, 1987; Palm, Swift and Woome, 1996) and higher soil moisture levels (Klinge *et al.*, 1998). Harcombe (1977) reported increases in carbon losses of 7–54% associated with the mineralisation of litter including fine roots upon forest clearing in Central America whilst Sanchez, Villachica and Bandy (1983) found increases in mineralisation rates of 25% for carbon and 17% for nitrogen upon forest cutting in the Peruvian Amazon. In contrast to the above findings, *reduced* mineralisation rates have been observed where the litter was rather dry, particularly where the litter layer had also become reduced, scattered or largely destroyed, e.g. during forestry operations or slash and burn cultivation (Ewel, 1976; Brouwer, 1996). Van Dam (2001) found net-nitrogen mineralisation in logging gaps of more than 200 m² to be four times lower compared to those in surrounding undisturbed forest on sandy soils in Guyana. On the other hand, Klinge (1998) measured 7–10 times higher nitrate concentrations in percolating soil water at 25 cm depth in eastern Amazonia soon after forest cutting (and before burning). These high values were attributed to rapid initial mineralisation of slash and, especially, decaying fine roots in the topsoil. Evidence for the latter comes from the fact that

Table 25.5. Relative nutrient losses due to volatilisation during burning of residual slash as a function of fuel mass and mass reduction

	Slash (t ha ⁻¹)	Reduction (%) ^a	Percentage nutrient losses					Reference
			N	P	K	Ca	Mg	
Costa Rica	38.5	83	23					Ewel <i>et al.</i> (1981)
Australia	11.4	64	64	44	55	33	37	Raison <i>et al.</i> (1985)
Brazil	6.3	90	95	51	44	52	42	Pivello and Coutinho (1992)
Fiji	40	86	84	52	79	78	60	Waterloo (1994)
Brazil	31.2	94	98	47	48	35	40	Mackensen <i>et al.</i> (1996)
	33.5	91	94	27	16	9	17	
	95.2	96	98	33	31	24	43	
Average		86	82	42	46	39	40	

^a Relative amounts of slash reduction added for comparison.

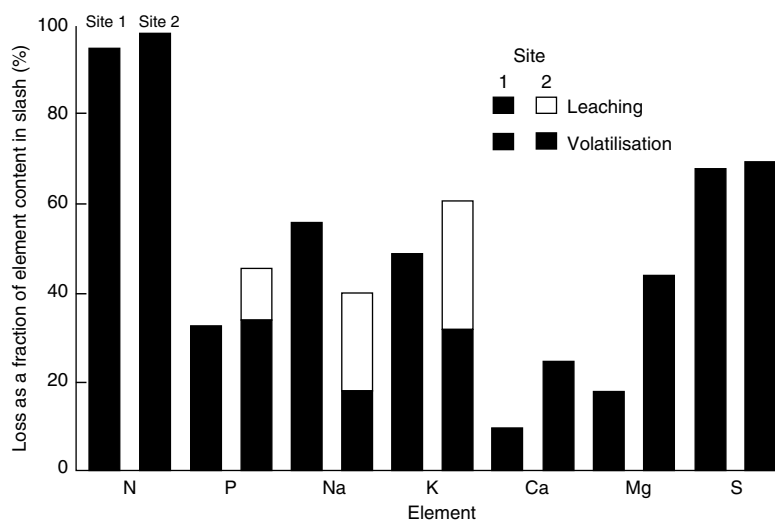


Figure 25.9 Nutrient losses upon burning of harvest slash (left histograms 33 t ha⁻¹ and right 90 t ha⁻¹) in a secondary forest in eastern Amazonia. Lower part of the bars indicates losses due to volatilisation

concentrations of calcium and magnesium in topsoil moisture were elevated after felling (and before burning) although these elements were not leached in detectable quantities from the residual slash (Klinge, 1998) (Figure 25.9).

Nutrient losses through slash burning

Residual phytomass is often burned on the site to facilitate access for planting. Volatilisation and ash particle transport during and shortly after burning can contribute significantly to overall losses of certain nutrients following site conversion to plantations (Figure 25.9, Table 25.5).

Reported atmospheric nutrient losses associated with burning range from 9–98%, the wide range reflecting differences in the fire intensity and volume of burned slash (Table 25.5). In the majority of studies the loss is closely related to the relative weight reduction

during burning, the upper part represents losses due to leaching from the slash before burning. All values given as percentage of the element content of the slash. (After Mackensen *et al.*, 1996.)

of the slash which is, in turn, a function of the intensity of the fire (Mackensen *et al.*, 1996). Losses of phosphorus (P), potassium (K), calcium (Ca) and magnesium (Mg) are generally comparable and average 39–46% whilst the average loss of the more volatile nitrogen (N) is 82% (Table 25.5).

The transformation of organic matter into mineral ash through slash burning has a most profound and usually rather immediate impact on soil fertility since mineral ash is readily soluble and easily washed into the soil (Khanna, Raison and Falkiner, 1994). Again, the more material that is burnt and the larger the amount of mineral ash left on the site, the higher the impact on soil fertility and the higher the potential for losses through subsequent leaching (Malmer and Grip, 1994; see also the next section). A comparison of soil nutrient characteristics before and after burning generally shows increases in soil pH and base saturation levels, possibly even

an increase in cation exchange capacity as well (Tomkins *et al.*, 1991). The extent of the rises in topsoil pH and base saturation is governed by initial soil acidity and the amount of ash. The duration of the effect differs strongly between sites as a function of rainfall regime (and thus leaching potential) and edaphic characteristics (notably soil texture and clay mineralogy; Sanchez, 1976). For example, Nye and Greenland (1964) reported an initial increase in pH from 5.2 to 8.2 in the top layer of an Alfisol/Luvisol in Ghana (moderate rainfall) after burning 50-year-old secondary growth. About two years later, the pH still amounted to 7. Conversely, only a modest increase in pH (from 3.8 to 4.5) was noticed after burning on a highly depleted Oxisol/Ferrasol under a more intense rainfall regime in Central Amazonia. In addition, the increase disappeared within four months (Brinkmann and Nascimento, 1973). Examples of changes in soil chemical conditions in the context of tropical plantation establishment include the plot-scale studies by Klinge (1998) in eastern Amazonia (Oxisols, covering one year) and, at the plantation scale, those by Ruhayat (1989) and Mackensen (1998) in East Kalimantan, Indonesia (both covering one rotation or 10 years; Ultisols/Acrisols).

Nutrient leaching

Clearfelling and burning of old-growth forest results in (at least temporarily) decreased evapotranspiration (Table 25.2) and thus in increased drainage through the soil profile and water yield from a catchment (Klinge *et al.*, 1998; Malmer, 1992; Grip *et al.*, this volume). This is accompanied by enhanced nutrient leaching from the site as illustrated by a small but growing number of studies (Uhl and Jordan, 1984; Malmer and Grip, 1994; Waterloo, 1994; Klinge, 1998).

Klinge (1998) used vacuum-tube lysimeters and recording tensiometers to determine leaching losses at 25, 40, 60 and 110 cm depths during the conversion of old-growth forest to a eucalypt plantation in eastern Amazonia. Concentrations of calcium, magnesium and nitrate ($\text{NO}_3\text{-N}$) in the percolating water showed a significant increase at 25 cm depth within two weeks after felling, whereas concentrations of potassium were raised only upon slash-burning. Although phosphorus was found to be leached from the residual phytomass (Figure 25.9), no increase in $\text{PO}_4\text{-P}$ concentrations in topsoil moisture was found, probably because of immediate phosphorus fixation in the topsoil (Uehara and Gillman, 1981). The observed increase in amounts of phosphorus stored in the topsoil after site conversion supports this idea (Klinge, 1998). Nutrient transfers to the subsoil were greatly enhanced upon the start of the rainy season when nutrients were rapidly leached beyond a depth of 1.1 m, and peaked about five months after forest conversion and burning (Klinge, 1998).

Brouwer (1996) used a similar (though non-automated) approach and observed comparable patterns of nutrient leaching in gaps of different sizes created by logging on sandy soils

in Guyana. Although no slash was burned, nutrient concentrations in soil water at 1.2 m depth peaked after about three months, reflecting the lower water and nutrient retention capacities of these sandy soils compared to the soils studied by Klinge (1998). In addition, the magnitude of leaching was shown to depend on the size of the canopy opening, with the largest losses being associated with the largest gap which also experienced the greatest soil disturbance (Brouwer, 1996). At a somewhat larger scale (headwater catchments of 3.4–9.7 ha underlain by a mixture of clayey and sandy soils) in Sabah, Malaysia, Malmer and Grip (1994) observed an almost immediate response of potassium concentrations in streamflow during storms after logging only and after logging followed by burning, with the greatest increases in the latter case. Concentrations of $\text{NO}_3\text{-N}$ took four weeks to respond. Much of the increased leaching losses occurred during stormflows in the form of shallow throughflow (SSF) and valley bottom saturation overland flow (SOF; Malmer and Grip, 1994; cf. Bonell, this volume). Despite the methodological differences between catchment and plot-based studies, differences in observed response times and process dynamics may be explained in terms of contrasts in precipitation and soil water (hillslope runoff) regimes and the soil exchange complex (clay and organic matter).

The interaction between percolating nutrients and the soil exchange complex is an important determinant of leaching patterns. For example, soils with 1:1-type clay minerals (kaolinite) and low organic matter content such as the Oxisols studied by Klinge (1998), retain less base cations than do Ultisols with their 2:1-type clay minerals. Under certain conditions, the 1:1-type clays also develop stronger bonds to potassium than to calcium (Levy *et al.*, 1988; Udo, 1978). This particular interaction probably accounts for the delayed peak in potassium concentration in the soil solution observed by Klinge (1998). During the period that concentrations of calcium in the soil solution peak, aluminium (and to a lesser extent hydrogen) will be replaced on the soil exchange complex and this leads to enhanced concentrations of Al^{3+} and H^+ in the soil solution (Klinge, 1998; Brouwer, 1996). This 'free' Al^{3+} in the soil solution reacts with H_2O to produce $\text{Al}(\text{OH})_3$ and thus contributes to a further increase in H^+ concentrations. Together with the enhanced rates of mineralisation and nitrification signalled earlier, this explains the temporary decrease of the pH in the soil solution down to a depth of 0.6 m at the Amazonian site of Klinge (1998). The pH in soil solution rises again after the calcium peak subsides. Similar drops in soil water pH were observed below gaps created by logging in the absence of fire in Guyana (Brouwer, 1996) and were also described by Sollins *et al.*, 1994.

The *duration* of enhanced nutrient leaching depends on site management (including soil disturbance), the above- and below-ground nutrient pools represented by the cleared vegetation, soil

texture (retention capacity) and the rate of growth of the new vegetation. In the eastern Amazonian example, increased nutrient concentrations in the deeper soil solution lasted for 6 to 9 months, with the exception of K and SO₄-S whose concentrations remained elevated down to depths of 110 cm for at least 12–15 months depending on the initial amount of slash that was burnt (Klinge, 1998). In East Malaysia, overall nutrient concentrations in baseflow remained elevated compared to a control stream for almost two years after logging or logging and burning, while the extra effect caused by the use of fire was detectable for about one year (Malmer and Grip, 1994). In Guyana, the accumulation of slowly decomposing slash in the largest gap (3440 m²) resulted in a comparatively small nutrient flux increase which remained significantly elevated for about 15 months, although concentrations of Ca, Mg, SO₄ and NO₃ remained slightly elevated for as long as 7 years after gap creation; Van Dam, 2001, cf. Figure 21.3 in Malmer *et al.*, this volume. The substantial surface disturbance by heavy machinery retarded the regeneration of the vegetation in the large gaps in this particular case but where regrowth is rapid and vigorous, nutrient losses will be reduced by accumulation in the phytomass. Brouwer (1996) observed reduced concentrations of K in percolating soil water compared to adjacent old-growth forest after about 34 months, indicating that by then the build-up of nutrient stocks may have begun. Uhl and Jordan (1984) reported a similar phenomenon for Ca and NO₃ during forest regrowth in the Venezuelan Amazon for a site with clayey soils about two years after felling and burning. Fast-growing tree species and dense undergrowth are equally helpful in this respect. Undergrowth in particular can play an important role in the short- to medium-term storage of nutrients (Smethurst and Nambiar, 1995). Cutting of undergrowth led to increased nutrient losses through leaching in Amazonia (Klinge, 1998) whereas a similar measure during the first years of plantation development in Indonesia resulted in cumulative nutrient losses which were quantitatively comparable to losses initiated by the initial site conversion (see Mackensen, 1998, for details). Summarising, enhanced leaching is a rather temporary process and although the associated nutrient losses can be substantial, they are usually much smaller than those associated with timber removal (Bruijnzeel, 1998; Mackensen, 1998; Figure 25.10a–d).

Erosion

Even in undisturbed forest ecosystems and tree plantations with a well-developed litter layer some erosion may occur. Reported rates in the early literature (summarised by Wiersum, 1984b) range from 0.02 to a maximum of 6.2 t ha⁻¹ yr⁻¹ (median values of 0.3 and 0.6 t ha⁻¹ yr⁻¹ for forests and plantations, respectively; $n = 47$). During the establishment phase, erosion is likely to increase significantly although the effect is usually short-lived

(Grip *et al.*, this volume; cf. Zhou *et al.*, 2001) (Figure 25.8). Wiersum (1984b) derived a median rate of 51 t ha⁻¹ yr⁻¹ for clean-weeded and otherwise disturbed and burned tree crops (range 1.2–183 t ha⁻¹ yr⁻¹; $n = 24$). The associated nutrient losses can be significant for the overall nutrient budget of plantation sites, especially on low-fertility soils and for stands grown in short rotations where such disturbances will occur every 7–10 years (Gonçalves *et al.*, 1997; Mackensen, 1998) (Figure 25.10a–d). However, nutrient losses due to surface erosion may also be important in plantations grown over longer rotations and on more fertile soils. In various densely populated countries where land pressure for food production is high (e.g. Indonesia), new plantations are often established through a *taungya* system in which local farmers are allowed to grow their crops between the newly planted trees in exchange for labour during plantation establishment. The cropping phase officially lasts three years, after which the trees and undergrowth take over and the farmers move to new fields within the forest estate. However, erosion during the *taungya* phase can be substantial (Wiersum, 1984b) and in one of the few assessments in the context of a plantation nutrient budget the associated nutrient losses were considered to be at least as high as the corresponding losses via timber harvesting, particularly when considering total rather than exchangeable nutrients contained in the eroded soil material (Bruijnzeel, 1992).

Declining soil nutrient reserves in intensively managed plantations

The nutrient enrichment of topsoils after rainforest conversion to pasture or cropping is a passing phenomenon (Hölscher *et al.*, this volume) and should not lead one to assume that the system as a whole is also enriched in nutrients. Rather, because of the high management-induced nutrient losses during plantation establishment described in the previous section (volatilisation, erosion, leaching), forest conversion is likely to contribute significantly to long-term depletion of fertility in the plantation system (Spangenberg *et al.*, 1996; Fölster and Khanna, 1997). As shown in Figure 25.10a–d, tropical forest conversion and intensive plantation management result in distinct negative nutrient fluxes. As a result, the newly established nutrient equilibrium will be at a lower level than the previous one under natural forest and continue to decrease with each subsequent rotation (Jordan, 1985). Therefore, continued nutrient exports through repeated biomass removal in subsequent rotations may well endanger sustainable plantation productivity, especially on poor sites (Oxisols/Ferralsols, Ultisols/Acrisols, Spodosols/Podsols) with the critical nutrient differing between situations (Spangenberg *et al.*, 1996; Fölster and Khanna, 1997; Mackensen, 1998; Nykvist *et al.*, 2000; Mackensen *et al.*, 2003). Sufficient compensation of nutrient losses through management-independent input fluxes such as precipitation or

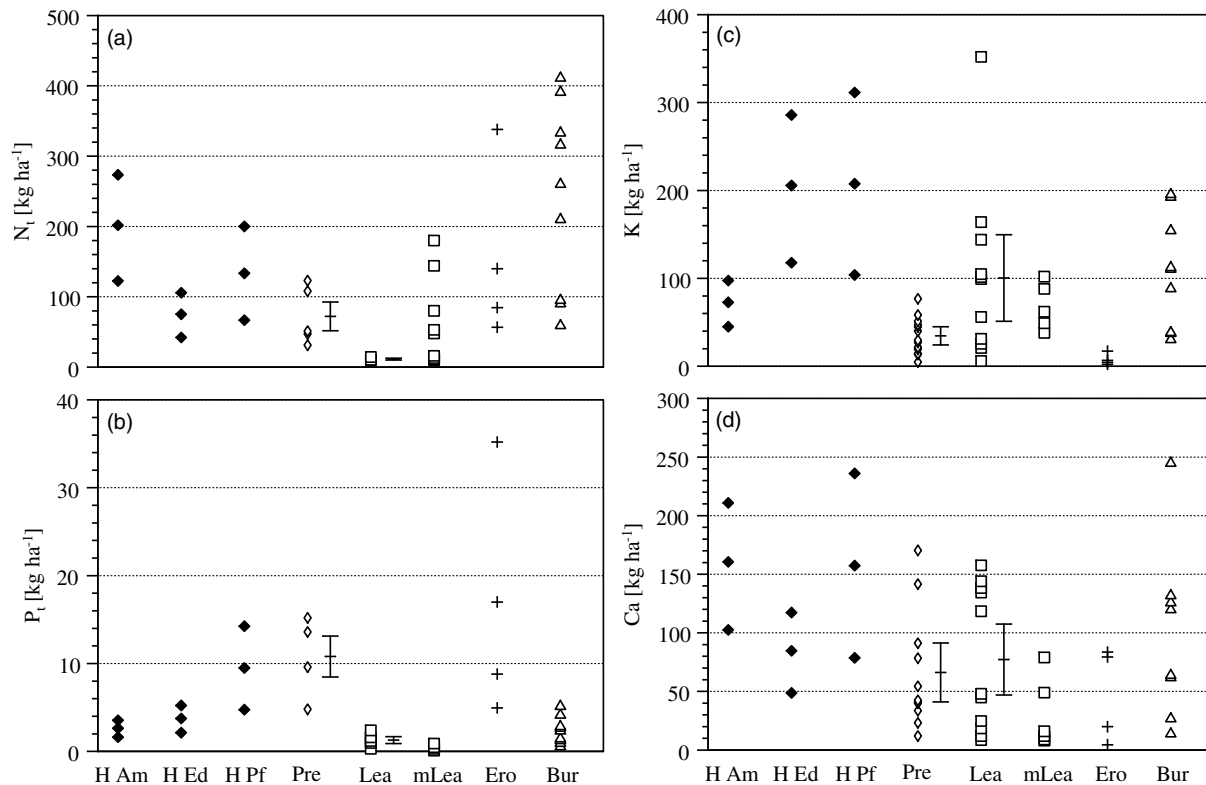


Figure 25.10 Comparison of nutrient gains and losses throughout a rotation of eight years for plantations of *Acacia mangium* (Am), *Eucalyptus deglupta* (Ed) and *Paraserianthes falcataria* (Pf) on Ultisols in East Kalimantan, Indonesia: (a) nitrogen, (b) phosphorus, (c) potassium, and (d) calcium. Explanation of remaining symbols: H Am, H Ed and H Pf – Nutrient losses associated with the harvesting of timber for the three different tree species and at harvesting volumes of 100, 200 and 300 m³ ha⁻¹ respectively; Pre – Nutrient gains through

weathering is unlikely, particularly where soils are highly weathered. Klinge (1998) calculated that nutrient inputs through bulk precipitation would take between 10 and 230 years to compensate for nutrient losses triggered by plantation establishment in eastern Amazonia. Similarly, periods of 35–55 years would be needed to cover losses of Ca, Mg and K associated with the clearance of rainforest to make way for *Acacia mangium* plantations in Malaysia on the basis of atmospheric nutrient inputs only. These were reduced to 4–20 years after inclusion of potential contributions by weathering (see Bruijnzeel, 1998, for details). It should be noted, however, that these calculations do not yet include any losses associated with the removal of timber at the end of the first rotation. At any rate, therefore, growing *A. mangium* on a ten-year rotational basis is bound to deplete the system beyond naturally occurring nutrient gains.

Because of the steady depletion of overall nutrient reserves under intensive tropical tree plantation schemes fertilisation would seem inevitable. However, as discussed in detail by Fölster and

precipitation (regional literature data); Lea – Baseline nutrient losses via leaching from natural forest (regional literature data); mLea – Management-dependent nutrient losses through leaching following stand conversion and preparation; Ero – Nutrient losses through erosion (assumed losses of topsoil of 50 and 200 tonnes ha⁻¹); Bur – Nutrient losses through burning of residual slash for harvest volume of 300 m³ ha⁻¹. (Modified from Mackensen, 1998.)

Khanna (1997) and Gonçalves *et al.* (1997), fertiliser application is not without problems either. Apart from economic considerations (Mackensen, Fölster and Ruhayat, 2000) there are dangers of acidification and exhaustion of (micro-)nutrients not covered by the application. Therefore, Fölster and Khanna (1997) advocate the harvesting of stem wood only and leaving bark material on-site where possible (cf. Crane and Raison, 1980; Bruijnzeel and Wiersum, 1985).

Conversion of grasslands into plantations

Studies of nutrient budgets following forestation of tropical grasslands are extremely rare. To make matters worse, the results of one of the very few ‘false time series’ studies available (Waterloo, 1994) are confounded by problems of different initial soil fertility levels between plots. Generally, the lower overall nutrient storage in degraded grassland ecosystems as compared to rainforests on similar soil types can be expected to lead to much lower

volatilisation and leaching losses upon plantation establishment (cf. Malmer and Grip, 1994). Similarly, such losses might also be compensated faster by nutrient inputs from bulk precipitation or (where applicable) weathering of soil parent material. Under subhumid conditions in the Southern Congo, savanna vegetation on low fertility sands has been converted to eucalypt plantations. Both the plantation and the trees in the savanna were found to be highly efficient in retaining nutrients from atmospheric inputs, with very little loss of plant-utilisable nutrients being leached out of the rooting zone (Laclau, Boillet and Ranger, 2000).

As is the case with natural forest recovery (Hölscher *et al.*, this volume), overall nutrient availability will determine the growth rate of forestation schemes in grasslands. Equally important is the intensity and duration of previous management (e.g. grazing on pastures) as these prove to be strong determinants of site nutrient status upon forestation. For example, in abandoned pastures in eastern Amazonia, Uhl, Buschbacher and Serrão (1988) observed above-ground biomass accumulation of secondary vegetation to be considerably higher on moderately used pastures than on intensively used pastures. Similar findings were reported by Buschbacher, Uhl and Serrão (1988), Hughes, Kauffman and Jaramillo (1999), Aide *et al.* (1995), and Fearnside and Guimaraes (1996). Furthermore, the extent of nutrient losses through leaching will depend also on the management techniques applied during tree planting. Strip-planting, which opens only a narrow strip of grass for tree planting is likely to result in smaller nutrient losses than the complete destruction of the grass cover through ploughing, slash burning or herbicide application (Otsamo *et al.*, 1994).

In the long run reforestation of degraded grassland sites may have positive effects on soil nutrient status. Over time plantation stands accumulate more nutrients and organic matter in the system than the previous grassland would and nutrient build-up during long rotations (>30 years) may match those of natural forest types on similar soils. As indicated previously, enhanced nutrient and organic matter cycling in the topsoil, nutrient pumping from deeper soils layers that were out of reach for the grass roots, and improved utilisation of nutrient input by precipitation (cf. Laclau *et al.*, 2000) may all contribute to improved fertility of reforested grasslands although hard evidence for this seems to be lacking at present (Waterloo, 1994; Young, 1997). Ironically, it is this same deep-rooting nature of trees that is partly responsible for their increased water use compared to grasslands or annual crops.

CONCLUSIONS AND RECOMMENDATIONS

Based on a review of a rather limited body of applicable research and partly extrapolating from hydrological studies in other parts

of the world, we conclude that the prospects for enhanced rainfall and augmented base flows as a consequence of forestation in the humid tropics are generally poor, but dependent on site-specific factors.

In terms of being able to predict the hydrological effects of land use changes it is especially important to understand the local hydrology, at least in terms of the dominant processes governing runoff generation. Erosion and sediment production from degraded landscapes may be readily moderated and stabilised at very low levels by forestation, provided they are not primarily determined by geological and topographic factors. In the longer term, the frequency of disturbance and the quality of forest management, particularly as it relates to roads and harvesting of timber and litter, will determine the on-going erosion risk and sediment yield of reforested catchments. Where rainfalls are particularly intense, and where soils are particularly clayey or degraded physically, there is greater potential for overland flow and near-surface throughflow to contribute to stormflows. In these situations there is the greatest opportunity for degraded catchments to be restored to improved hydrological function through forestation. Where soils are deep and porous and comparatively little disturbed, the effect of forestation on stormflows will be modest and more pronounced through lowered baseflows.

The establishment of plantations leads to major changes in the characteristics and nutrient fluxes of the ecosystem. Whether these changes degrade or improve soil fertility depends on the original land cover (forest or degraded grassland) and the associated management practices. Forestation of degraded land by appropriate management practices may result in a general improvement of soil fertility (although hard evidence from the humid tropics seems to be lacking), while the replacement of native forest by plantation generally initiates a range of processes resulting ultimately in a decline in soil fertility. The nutrient enrichment of topsoils after forest conversion is only very temporary and does not represent higher levels for the ecosystem as a whole. Rather, the high management-induced losses of nutrients during plantation establishment and harvesting (leaching, volatilisation, erosion) contribute significantly to overall nutrient depletion. The newly established nutrient equilibrium will be on a lower level than the previous one with each subsequent rotation.

There are very large areas of land being planted to trees in the humid tropics and given the relative paucity of direct research on the hydrological effects of this change in land use, there is a high priority for studies in this particular field. The following aspects are considered to be particularly important:

- Studies need to measure factors other than just those directly related to the vegetation change, such as effects on soil hydraulic properties which may be as important as the

changes in water use and soil moisture levels imposed by forestation.

- The hydrology of degraded catchments needs to be described in terms of real measurements of actually occurring processes as too much depends on assumptions in the existing literature.
- It is suggested as a working hypothesis that in the conversion from degraded grasslands to timber plantations, the increases in infiltration that can be attributed to a forest cover are likely to be exceeded by the increase in the evaporation component of the water balance of the new forest. Research should determine under which specific situations this hypothesis does or does not apply.
- There is a need to include groundwater in the study of changed catchment water balances associated with forestation, especially as the socially important baseflows and dry season flows are normally generated from groundwater stores.
- It is expected that critical variables in determining the response of (degraded) catchments to forestation will be the nature of the rainfall and the soil properties. These aspects ought to be thoroughly measured and documented.
- All the direct and indirect hydrological effects of a major land use change will not be visible in the short term, and there is therefore a need for properly controlled long-term studies, such as paired catchment experiments backed up by process-based research.
- Knowledge of plantation nutrient dynamics and fertility over entire forest rotations requires holistic studies that are broad enough in both time and space, as well as in terms of ecological scope.

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