Chapter 22
Tropical forest degradation and aquatic ecosystems: our current state of knowledge

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Abstract

Recently available data illustrate that tropical forests are being degraded over very large spatial scales. Estimates indicate that tropical countries are losing 127 300 km^2 of forest annually, a further 55 000 km^2 of tropical forests are logged each year, and while it is difficult to quantify the amount of tropical forest that is burnt each year, it may be as much as 30 000 km^2. Conservation agencies have raised interest in addressing deforestation impacts; however, the vast majority of the efforts have been directed towards understanding and mitigating impacts to terrestrial habitats. In the tropics, the impacts of forest degradation on aquatic systems have received little attention. Available evidence on the impacts of tropical forest degradation on aquatic systems, including changes in flood regimes, siltation, sedimentation, removal of overhanging vegetation, and changes in invertebrate and fish communities, are reviewed.

Keywords: deforestation, fire, fish, logging, siltation, tropical aquatic systems.

22.1 Introduction

Accelerating deforestation and forest degradation in the tropics is generating widespread concern throughout the world. Conservation agencies have raised awareness and interest in addressing deforestation impacts; however, the vast majority of the efforts have been directed towards understanding and mitigating impacts to terrestrial habitats. In the tropics, impacts of deforestation and forest degradation on aquatic systems are largely unknown. Yet, such ecosystems are extensive and support an extremely high richness of freshwater fishes. For example, the Congo River is 4650 km long, its drainage basin encompasses 3.8 million km^2, and it supports more than 560 species of fishes. Similarly, the Amazon River is 5500 km long with a drainage of 5.6 million km^2 and supports over 1300 fish species (Chapman 2001). This biodiversity is easily jeopardised by severe perturbation (Chapman, Chapman, Ogutu-Ohwayo, Chandler, Kaufman & Keiter 1996). For example, Mohd (1994) documented that 41% of the native fish species in the Gambak River of Southeast Asia were lost between 1969 and 1990 and attributed this to logging, highway construction, and land clearing for agriculture. Millions of people depend on these aquatic systems for food,

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drinking water, transport, etc. For example, it is estimated that 30 million people depend on the fisheries of Lake Victoria in some capacity (Kaufman 1992; Kaufman, Chapman & Chapman 1997).

Here the patterns of deforestation and forest degradation in the tropics are reviewed and the potential consequences of these events on inland waters are examined. Wherever possible case studies conducted in the tropics are presented but, due to the paucity of information on tropical systems, potential consequences of forest degradation are also illustrated by reference to studies conducted in temperate regions or through speculation. The process of conducting this review illustrates the current state of knowledge of the impacts of tropical forest degradation on tropical aquatic systems, and suggests a number of areas where further study is required.

22.2 Tropical forest degradation

Current data on worldwide forest cover are provided by FAO (1999), making it possible to estimate the magnitude of impact that deforestation and forest degradation will have on aquatic systems. For developing countries the FAO defines deforestation as the depletion of tree cover in closed-canopy forests to less than 10%, a canopy thinning threshold that will likely have significant impacts on neighbouring water bodies. In 1995 there were 19 555 700 km^2 of forest remaining in tropical countries (FAO 1999). Forest loss between 1980 and 1995 was 10.5% for Africa, 9.7% for Latin America and the Caribbean, and 6.4% for Asia and Oceania. Tropical countries lose 127 300 km^2 of forest annually; an area greater than Mississippi (122 335 km^2) or just smaller than Greece (131 985 km^2). The highest losses occurred in countries with large expanses of tropical forest, including average annual conversions of 25 540 km^2 for Brazil, 10 840 km^2 for Indonesia, and 7400 km^2 for the Democratic Republic of Congo (Fig. 22.1). The top four countries losing the greatest proportion of their remaining forest cover are the Philippines (annual deforestation rate of 3.87%), El Salvador (3.81%), Costa Rica (3.29%) and Sierra Leone (3.28%).

Tropical deforestation appears to be driven primarily by frontier expansion of subsistence agriculture and large economic development programmes involving resettlement, agriculture and infrastructure (FAO 1999). Growing external debts place strong pressures on governments to encourage timber harvesting and increased agricultural activity. For example, each year the countries of sub-Saharan Africa return a mean of 58% of their gross national product (GNP) in repayments of foreign debts and the repayment can be as high as 241% of the GNP (Stuart, Adams & Jenkins 1990).

According to the FAO definition, selective logging is not considered deforestation, since it does not decrease forest cover to less than 10% of its original level. Yet, selective logging operations can dramatically impact aquatic systems (Pringle & Benstead, 2001). Currently, it is estimated that between 50 000 and 60 000 km^2 of tropical forests are logged each year, approximately a third of the area that is completely deforested (FAO 1993).

With the proliferation of forest fires throughout Southeast Asia (Kinnaird & O'Brien 1999) and South America (Nepstad, Veríssimo, Alencar, Nobre, Lima,

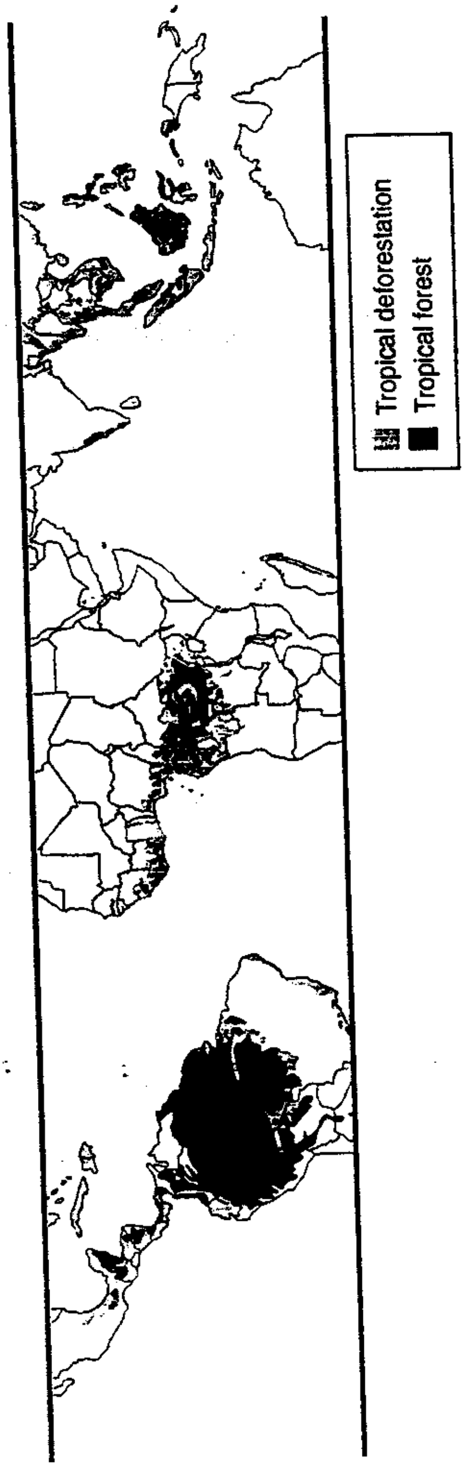

Figure 22.1 Map of the world illustrating the major regions of moist and wet forest and the extent of deforestation in these areas (adapted from National Geographic Atlas 1992)

Lefebvre, Schlesinger, Potter, Moutinho, Mendoza, Cochrane & Brooks 1999), and the media coverage that they have incited, it is increasingly evident that wildfires are having significant impacts on tropical ecosystems. The prevailing opinion concerning fire ecology in tropical forests is that natural fires are relatively rare, and that the majority of fires seen today are either induced or aggravated by human activity (Janzen 1986; Tutin, White & Missandzouo 1996). For example, when logging operations leave an area, large amounts of deadwood are left behind. Since, the canopy has been opened, this wood dries and becomes a large source of flammable material.

Determining the amount of tropical forest recently burned using conventional satellite imagery is at best difficult (FAO 1999) because many fires are restricted to the understory, and leave much of the canopy relatively intact (Nepstad *et al*, 1999; Peres 1999). Obtaining representative figures for the amount of tropical forest that burns annually is further complicated because there is large year-to-year variability in the extent of fires, which are primarily mediated by supra-annual El Niño events. Despite these difficulties in quantifying the amount of tropical forests burned each year, the magnitude of forest fires can be illustrated by presenting a couple of examples. FAO (1999) estimated that in 1997 and 1998 an area of 2 million ha of forest burned in Brazil and 4 million ha burned in Indonesia. Between December 1997 and April 1998, more than 13 000 fires burned in Nicaragua destroying vegetation on more than 800 000 ha of land. These estimates appear to be conservative since at least 1 million ha of intact forests burned in the State of Roraima (Brazil) alone following the 1997/1998 El Niño dry season (Shimabukuro, Krug, Santos, Novo & Yi 2000). At this time, almost half of the forest cover in the entire Brazilian Amazon (1 550 000 km^2) had already completely exhausted its ground-water supply to a depth of at least 10 m, and were therefore highly inflammable (Nepstad *et al.* 1999).

If one made a conservative estimate that 30 000 km^2 of forest burns each year, then the amount of forest deforested, logged, or burned is approximately 212 000 km^2. While conservation agencies have raised widespread concern about this extensive degradation of tropical forests, the impacts of this degradation on aquatic systems has been largely ignored.

22.3 Impacts of deforestation on aquatic systems

22.3.1 *Siltation and sedimentation*

An effect of deforestation readily apparent is increased siltation and sedimentation. There are many ways to remove logs from a logging site. Ground-based timber yarding is the predominate approach in the tropics, and involves felling widely separated trees and hauling out the sawn timber or logs. When trees are cut on site (e.g. pitsawing) and the timber is hauled out manually, there is little damage to neighbouring trees or to the soil (Chapman & Chapman 1996). However, more typically, tractors or bulldozers are used to haul out whole logs. When this is done roads must be constructed to allow machinery access. Logging with bulldozers in Brazil that removed only 2% of the trees greater than 10 cm diameter at breast height (DBH), resulted in damage to

26% of the remaining trees, canopy opening of 50%, and the development of logging roads covering 8% of the forest floor (Frumhoff 1995; Johns, Barreto & Uhl 1997). Uprooted trees and logging roads expose the soil to the rains and create situations where sediment readily runs off of the land into adjacent aquatic systems.

Deforestation of the watershed leads to changes in the seasonal flood regime. In forested watersheds, vegetation and topsoil help retain water. With their removal, flood peaks tend to become higher and shorter, because run-off is not delayed by the holding capacities of the forest. Dramatic changes in the flood regime can negatively impact fish populations that require a smoother seasonal transition (Welcomme 1985). The felling or poisoning of 40% of the upperstory trees in South America corresponded to a 55–70% increase in the water yield (C. Bruijnzeel, unpublished data). This increase can last for several years, since the regrowth that comes after logging possesses a smaller total leaf area that does not intercept rainfall to the same extent of mature vegetation (Uhl & Jordan 1984), and this secondary vegetation has a less-developed root system that cannot exploit soil moisture levels to the extent of mature forest trees (C. Bruijnzeel, unpublished data).

Removal of vegetation decreases the evapotranspiration rate and the interception of precipitation, and therefore increases runoff and sediment yield. In many tropical forested and savanna rivers, annual sediment yield is low. Sedimentation increases dramatically in the forested rivers following deforestation (Fig. 22.2). Increased turbidity

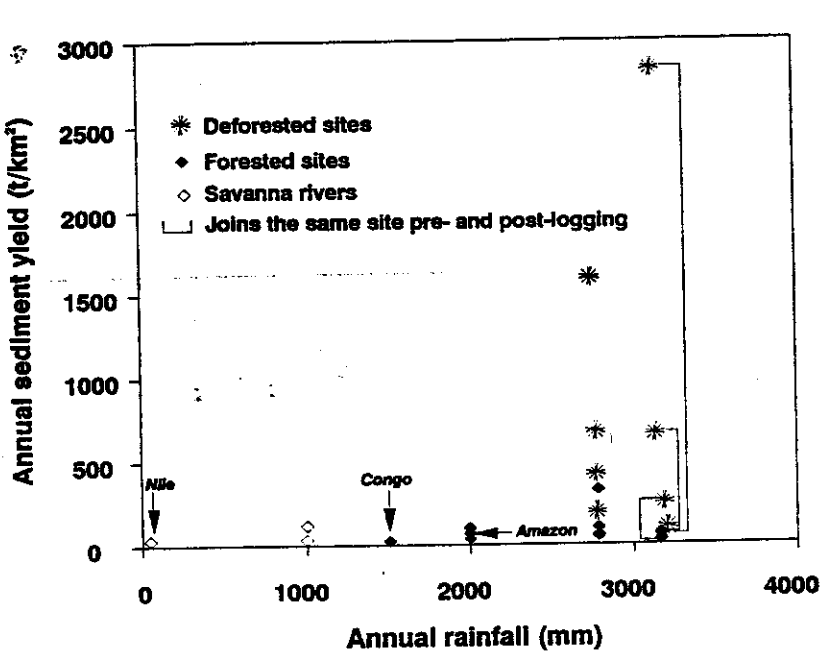

Figure 22.2 Annual sediment yield (t km^{-2}) for forested, savanna, and deforested catchments in the tropics *v.* annual precipitation for the region. Data abridged from Holeman (1962), Welcomme (1985) and Douglas *et al.* (1993). A vertical line joins sites for which data were gathered pre- and post-disturbance. All deforested sites are from Peninsular Malaysia

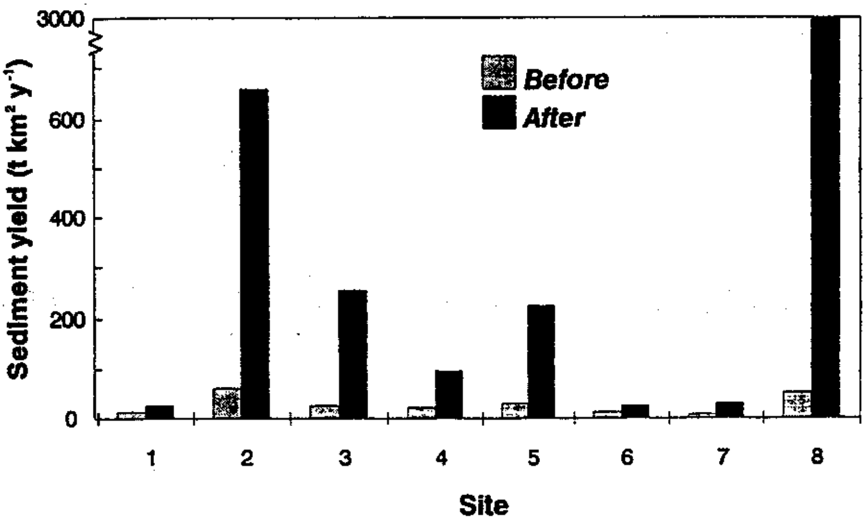

Figure 22.3 Sediments yields from rainforests in Malaysia and Kalimantan before and after logging (data from Kasran 1988; Douglas *et al.* 1993; Kasran & Nik 1994)

is associated with increased sedimentation and siltation, and much of the increased silt load of tropical rivers in recent history has been associated with the deforestation in the upper regions of the watershed (Marlier 1973; Eckholm 1976).

The most detailed tropical studies of how logging effects sediment loads in rivers have been conducted in Malaysia and Kalimantan (Fig. 22.3). A number of these studies also evaluated the degree to which reduced-impact logging decreased the amount of sediment coming off of logged areas. Reduced-impact logging typically involves removing the same amount of timber from an area as conventional logging, but it decreases forest damage by closely supervising tree felling, cutting vines before felling, by minimising road construction, and occasionally by cutting cross drains on steep logging roads. For a dipterocarp forest in Peninsular Malaysia, sediment yield increased from pre-logging levels by 97% for conventional logging and 70% for reduced-impact logging Kasran (1988). Yusop and Suki (1994) showed that relative to a control site in the Berembun Forest Reserve, Peninsular Malaysia, conventional logging increased turbidity by 9 times and suspended soil by 12 times. Reduced-impact logging only resulted in a two-fold increase in these characters. At a different study site, Kasran and Nik (1994) found suspended sediment yield increased 177% the year following logging and further increased 297% in the following year. Four years after, with the growth of secondary vegetation, the yield returned to the pre-logging level. Douglas, Greer, Bidin and Silsbury (1993) documented that logging or ground clearance increased river sediment yields by 2–50 times in Malaysia.

These studies in Malaysia are the type of research that is needed, because they not only assess the nature of perturbation that logging causes to aquatic systems, but also evaluate possible management options. However, it is questionable just how appropriate it is to use sediment-loading studies conducted in Southeast Asia as models for activities in other parts of the tropics. First, studies have shown a high degree of variation in the amount of sediments coming off of logged areas depending on the slope, the number and extent of road construction, and the underlying geology (Douglas *et al.* 1993). Second, much of the sediment released from logged areas can be deposited during very brief storm events, illustrating the need for intensive monitoring over a number of years. Unfortunately, long-term studies of this nature are rare. Finally, and possibly most importantly, the nature of the extraction differs among tropical regions. For example, African forests are very heterogeneous in their species composition. Consequently, harvesting activities produce a comparatively low volume per hectare compared to Southeast Asia. Yields from selectively logged forests in Africa are generally not more than $15\,m^3\,ha^{-1}$, while $60\,m^3\,ha^{-1}$ are commonly extracted from Southeast Asian forests (Bakouma & Buttoud 1996).

In a general survey of watersheds in Kenya, Dunne (1979) demonstrated that forested catchments lose $20–30\,t\,km^{-2}\,year^{-1}$. Sediment yields of agricultural lands vary enormously with runoff. In the wettest, steepest cultivated catchments, the soil loss exceeds $4000\,t\,km^{-2}\,year^{-1}$. The sediment yields from rangeland catchments are also variable, the driest catchments lose less than $100\,t\,km^{-2}\,year^{-1}$, while $20\,000\,t\,km^{-2}\,year^{-1}$ are exported from the wettest steepest grazed catchment.

22.3.2 *Changes in the biota and cascading effects*

Surveys of aquatic biota in tropical waters affected by different types of human habitat modification are few; however, researchers are becoming increasingly interested in using macroinvertebrates as indicators of stream 'health' (Fore, Karr & Wisseman 1996; Richards, Johnson & Host 1996; Stone & Wallace 1998). Changes in the physical environment associated with human modifications to the landscape, such as changing the amount of silt in the water, or changing light level, will impact the aquatic biota. These changes are complex, difficult to predict, and there are too few studies available from tropical areas to make generalisations that are sufficiently reliable to use in management. In Oregon, USA, Carlson, Andrus and Froelich (1990) documented that macroinvertebrate density was 20–113% greater at logged sites relative to unlogged sites, despite similar diversity between the logged and undisturbed sites. In contrast, sand deposition resulting from stream diversion in Cornwall, UK, was associated with a dramatic reduction in aquatic plants, the elimination of several species, and a lower diversity of macroinvertebrates (Nuttall 1972). Stone and Wallace (1998) found that 16 years after logging in the southern Appalachian mountains of the USA, benthic invertebrate abundance was three times higher in the disturbed stream.

Increased sedimentation and higher turbidity can lead to the decline of plankton through a reduction in light penetration, and the disappearance of many benthic, rheophilic animals that are sensitive to mud on their integument and gills or lose their interstitial habitats to clogging by silt (Marlier 1973; Welcomme 1983). These include insect groups like the Ephemeroptera, Plecoptera, and Trichoptera. Severe forest modification can result in shifts to invertebrate communities dominated by small burrowing forms, such as larval chironomids. Such burrowing forms may be less available to foraging fishes (Pringle & Benstead 2001). The silt can also provide anchorage for vegetation that can block low order streams (Welcomme 1983). On the floodplain, excessive siltation can choke the standing waters that then disappear at a rate faster than they are created.

In their review of the effects of logging on tropical riverine systems, Pringle and Benstead (2001) found no studies that dealt specifically with effects of increased sedimentation on tropical fishes. However, in temperate systems (focusing primarily on salmonids) negative effects include: smothering of eggs and entrapment of fry by consolidated sediments; mortality of juvenile and adult fish due to clogging of opercular cavities and gill filaments; and sub-lethal effects such as reduced feeding and growth, respiratory impairment, reduced tolerance to disease and toxicants, and physiological stress (Lloyd 1987; Waters 1995). A serious impact is also the loss of habitat for fry that require interstitial space in riffles, and the filling of pools and blanketing of structural cover (Waters 1995). Garman and Moring (1993) documented a decline in abundance of one fish species following clearcut logging and a decline in aquatic prey resources, while a second species increased in abundance because it switched to eating terrestrial arthropods. In a logged Eucalyptus forest in Tasmania, Davies & Nelson (1994) found that a significant decrease in riffle macroinvertebrates, particularly stoneflies and mayflies, corresponded to a decrease in brown trout, *Salmo trutta* L., abundance. Goulding, Smith and Mahar (1996) commented that floodplain deforestation in the Amazon resulted in a dwindling amount of wood that is naturally carried downstream.

Given that many rain forest rivers have relatively low primary productivity and rely largely on allochthonous input, destruction of the forest may result in a decline in aquatic productivity (Davies & Nelson 1994). In areas of the central Congo basin and extensive areas of the Amazon basin where flooding allows fishes to find food, refuge and breeding areas in the inundated forest, the consequences of deforestation may be particularly severe. Although data are few on the impact of flooded forest conversion in tropical rain forests, the recent history of the Mekong system provides a basis on which to speculate. In Cambodia, the Mekong River floods into the Grand Lac, a seasonally fluctuating lake, and surrounding floodplain (formerly forest). Many fishes move into the Grand Lac as it fills and then move out into the inundated zone, returning to the lake and migrating down river as waters fall in the dry season (Lowe-McConnell 1975). Chevey and Le Poulain (1940) found that growth of some cyprinids was faster in the flooded forest than in the open lake and the river that the authors attributed to abundant fish food in the forest. Subsequently, the forest surrounding the Grand Lac area of the Mekong was cleared for agriculture and firewood. This clearance was accompanied by a decline of about 50% in the fishery in 25 years (Welcomme 1985). The decline was attributed to erosion and siltation in the Mekong basin and the dramatic decline of allochthonous food arising from the deforestation. Siltation led to increased turbidity of waters, which in the case of the Grand Lac coincided with a drop in primary production reflected in decreased fish production. Welcomme (1985) reported that the process was reversed during the late 1970s when political troubles led to a collapse of the human population in the region, which permitted the forest around the Grand Lac to regenerate. Forest regeneration coincided with an increase in fish populations, which may have related to the renewal of forest resources and reduced fishing pressure. Similar effects have been observed in the Amazon basin, where the economically important characid, *Colossoma macropomum* Cuvier (the tambaqui) declined near the major city of Manaus. The decline of the frugivorous tambaqui was attributed to overfishing and the disappearance of the inundated forest feeding grounds (Marshall 1995). Welcomme (1983) noted that in heavily forested systems, such as the Amazon, Congo and Mekong, trees appear to act as a nutrient sink and their disappearance from the fundamentally poor systems may have long-term impacts on nutrient balance.

Cohen, Bills, Cocquyt and Caljon (1993) studied the effects of sedimentation on the species richness of three taxonomic groups in Lake Tanganyika: ostracods, fish and diatoms, and by contrasting undisturbed, moderately disturbed, and highly disturbed areas. The most serious threat facing Lake Tanganyika is viewed to be deforestation and subsequent erosion (Cohen *et al.* 1993). In the north end of the lake where deforestation approaches 100% of the forested land, rates of soil erosion range between 28 and 100 t ha^{-1} year^{-1} depending on the slope (Cohen *et al.* 1993). The species richness of ostracods and fish appeared to be heavily affected by sediment, while diatoms showed little change in species richness (Cohen *et al.* 1993). In a later study of the same region, Alin, Cohen, Bils, Gashagaza, Michel, Tiercelin, Martens, Coveliers, Mboko, West, Soreghan, Kimbadi and Ntakimazi (1999) showed that the species richness and density of fishes, molluscs and ostracods were negatively correlated with sedimentation.

22.3.3 *Effects of removing overhanging trees*

When one thinks of tropical riverine systems, one typically thinks of rivers like the Amazon, Congo, Mekong or Nile. However, many rivers are much smaller. For example, more than 90% of Africa's rivers are less than 9 km long (Stiassny 1996). As a result, many tropical river systems are strongly influenced by overhanging terrestrial vegetation. Removing trees bordering rivers causes a dramatic shift in sources of production, reduces the input of terrestrial elements (e.g. leaves, terrestrial invertebrates), increases the light and water temperature, and is associated with changes in the chemical composition of the water. Increased light resulting from the removal or loss of the canopy leads to higher standing stocks of algae (Ulrich, Burton & Oemke 1993). Such changes can restructure the macroinvertebrate communities. For example, in the southern Appalachian mountains, Wallace and Gurtz (1986) found that Ephemeroptera production was 28 times higher in a logged catchment in comparison to an unlogged steam. Presumably, this was the result of an increase in primary productivity associated with the opening of the canopy and increased light levels. Marlier (1973) reported changes in chemical composition associated with increased solar insolation for small streams in the eastern Congo. Waters from deforested areas had higher pH and higher concentrations of salts than protected waters.

The importance of maintaining riparian vegetation is highlighted in studies that have examined the value of maintaining a riparian buffer zone as a strategy to decrease effects of logging on stream habitats. Growns and Davies (1991) contrasted the lotic macroinvertebrate community of logged streams, a stream with a 100-m buffer zone, and control streams. Macroinvertebrate community structure in the buffered stream differed from that in the undisturbed stream, but it was more similar to the latter than to the logged stream. These results suggest that a 100 m wide buffer zone is effective at ameliorating disturbance due to clearfelling (see also Richards *et al.* 1996). Davies and Nelson (1994) documented the effects of logging for both cable and conventional logging in areas with a range of riparian buffer strip widths (0–50 m) and found that all impacts were significant only at buffer widths of less than 30 m.

22.4 Conclusions

The recently available information on deforestation, logging, and fire illustrates that tropical forests have been seriously impacted over very large spatial scales. The new millennium brings an even greater potential for change. The severity of this situation has been widely recognised for the last three decades. However, recognition of impacts to tropical terrestrial systems has not lead to sufficient effort to quantify the impacts of tropical forest degradation on aquatic systems, nor to many major conservation efforts to protect fresh waters. The future offers a great opportunity for research to contribute to freshwater aquatic conservation and management.

Research priorities should include:

(1) documenting patterns of change to terrestrial systems and associated responses of aquatic systems;
(2) understanding cascading effects of forest removal;
(3) predicting how different functional guilds (fishes, benthic invertebrates) will be affected by different types of human activities;
(4) identifying mechanisms determining the pattern of change.

Studies that attempt to address these research priorities should be set in a framework such that concrete management recommendations can result from the research (e.g. document patterns of change that evaluates different conservation approaches). Since the protection of entire aquatic systems is so difficult (e.g. protection of an entire drainage), one of the most rewarding avenues for future tropical research will be to determine simple mechanisms to minimise impacts that degradation of forests will have on neighbouring aquatic systems.

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