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Deforestation in Tropical Africa

Impacts on Aquatic Ecosystems

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There is widespread concern about expanding and accelerating deforestation in the tropics. Conservation agencies have raised interest and awareness in addressing deforestation impacts; most efforts, however, have been directed toward understanding and mitigating impacts to terrestrial habitats. In the tropics, the impacts of deforestation and land conversion on aquatic systems are largely unstudied, and the topic has received little attention from conservation organizations, managers, and local governments. Yet, such ecosystems are extensive. For example, the Congo River is 4,667 km long, and its drainage basin encompasses 3.7 million km^2. Millions of people depend on these aquatic systems for food, drinking water, and transport (e.g., 30 million people depend on Lake Victoria in some way) (Kaufman 1992, Kaufman et al. 1997). Furthermore, these systems harbor a reservoir of biodiversity easily lost if they are inappropriately treated. 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; he attributed the loss to a combination of logging, highway construction, and land clearing for agriculture.

In this chapter, patterns of deforestation and forest protection in Africa are documented, and potential consequences of these events on inland waters are considered. Whenever possible, studies conducted in Africa are presented; however, the paucity of information forces the use of studies in other tropical systems or in temperate systems. Whenever possible, the reliability of using studies from elsewhere as models for what may occur in Africa is discussed.

The objectives of this chapter are to first review trends of deforestation in Africa and, in doing so, to illustrate where there is the greatest need for concern and where the greatest potential conservation gains can be made. Second, the parks system in Africa is examined and whether park design has considered aquatic systems is discussed. Third, a discussion is presented of how particular types of land management affect aquatic systems. Finally, recommendations that attempt to connect aquatic and terrestrial conservation and management in Africa are made.

Deforestation Trends in Africa

It is estimated that the rain forests of Africa covered 3,620,000 km^2 before agricultural clearing and habitat alterations by people. Seventy-four percent of this area was found in Central Africa, 19% in West Africa, and 7% in East Africa (Martin 1991) (Table 11.1 and Figure 11.1). Although estimates from different sources vary considerably, it is clear that the area of forest remaining today has been drastically reduced. One estimate suggests that the amount of forest remaining is approximately 1,490,000 km^2, or 55% of the original area in Central Africa; 190,000 km^2,or 28% of the area in West Africa; and 70,000 km^2, or 28% of the original area of East Africa (Martin 1991). Food and Agriculture Organization (FAO) statistics estimate that in 1985 West African forests encompassed 143,260 km^2; Central Africa forests included some 1,717,450 km^2; and East African forest encompassed only 30,000 km^2. Recent satellite imagery indicates that the total area of closed tropical forest in Africa is between 1,850,000 and 2,150,000 km^2 (Mayaux et al. 1998); 37,000 km^2, or one-fourth of the area of Florida, is cleared annually (Boahene 1998). Although these estimates show discrepancies and are open to debate, it is clear that African forests have been dramatically reduced. The most extensive forested areas were found in West and Central Africa. However, deforestation has been most extensive in West African countries, particularly in Nigeria and Ivory Coast (Table 11.1). In contrast, the Central African forests are extensive, human populations are sparse, and deforestation rates have been low (Table 11.1; see also Table 9.2). East African countries have little forest, but some countries still have relatively high deforestation rates (e.g., Rwanda, Burundi). Most deforested areas have been converted to some other type of land use, typically agricultural or pastoral. Millions of hectares of forests have also been degraded by logging but are still classified as forest.

Barnes (1990) constructed a predictive model based on absolute area deforested per year to predict the extent of forest that will remain in decades to come. Changes in forest area predicted by this model are dra-

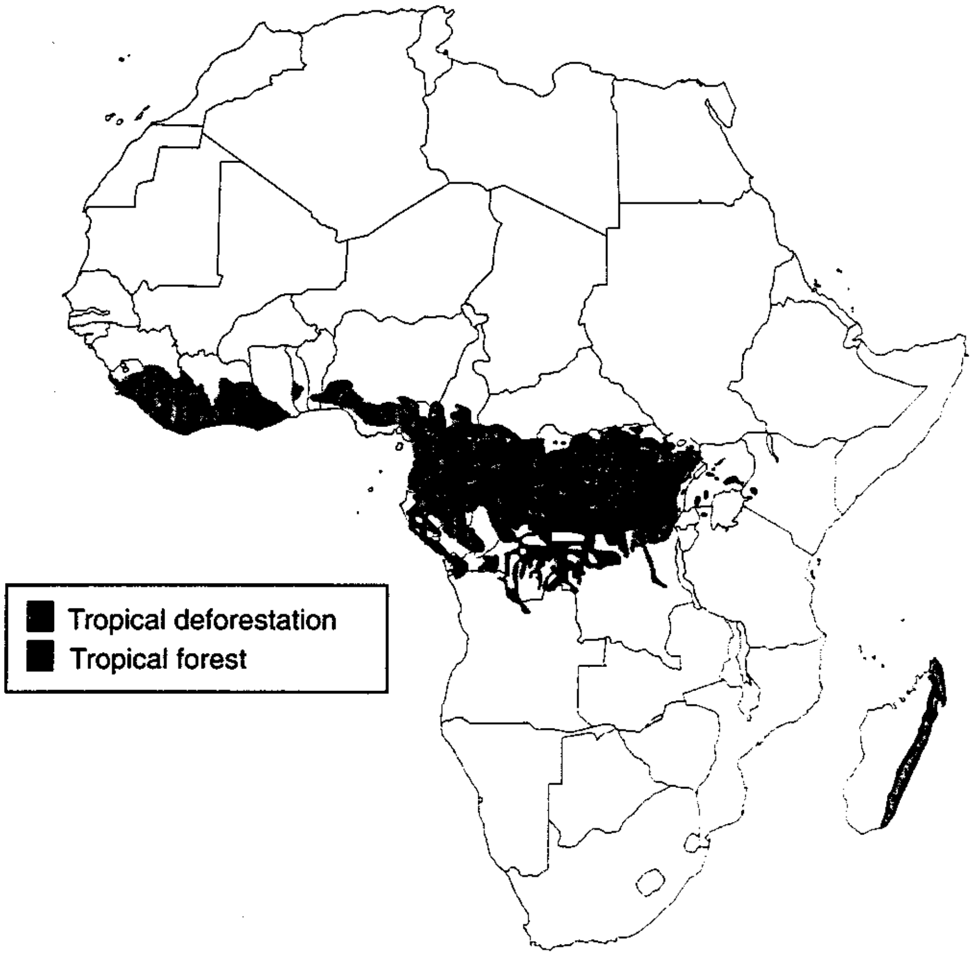

Fig. 11.1. Map of the African continent, illustrating the major regions of moist and wet forest and the extent of deforestation in those areas. (Adapted from *National Geographic Atlas of the World* 1992.)

matic. In West Africa the model predicts that 70% of the forests will be converted by the year 2040 (see Table 11.1). By that time, East African forests will have lost 95% of their area. It is only in Central Africa that large tracks of forest will remain, but even the extensive forest of the Democratic Republic of Congo is predicted to be reduced by 30%. This model does not consider reforestation efforts; however, the exclusion of reforestation efforts seems justified given the current extent of reforestation in Africa. Between 1981 and 1986 the total area that was reforested in 37 African countries was only 1,260 km^2; during that same period, 13,310 km^2 of forest were cut down in those same countries. Struhsaker (1981) estimated that for Uganda, the amount of land replanted (typically ble 11.1. Descriptive statistics and deforestation rates for African countries between 1976 and 1980

ntry	Size (km²)	Forested area (km²)	Mean annual percentage deforested	Population density (no. of people/km²)	Annual growth rate (%)	Projected area remaining by 2040 (km²)
t Africa						
Senegal	197,000	2,200		29.0	2.90	109
Gambia		6,500	1.08	55.2	3.65	214
Guinea-Bissau	36,000	6,600	2.27	21.3	2.20	4,232
Guinea	246,000	20,500	1.76	22.1	3.12	9,694
Sierra Leone	72,000	7,400	0.78	48.8	2.75	2,848
Liberia	111,000	20,000	2.05	16.8	3.30	12,419
Côte d'Ivoire	322,000	44,580	6.95	25.6	3.83	20,015
Ghana	239,000	17,180	1.57	48.3	3.10	3,598
Togo	57,000	3,040	0.66	45.4	3.22	733
Benin	112,620	470	3.19	30.9		0
Nigeria	394,000	59,500	4.79	91.7	3.17	1,548
tral Africa						
Cameroon	475,000	179,200	0.45	17.8	3.48	117,156
Equatorial Guinea	28,000	12,950	0.19	13.6	2.60	10,341
Gabon	265,000	205,000	0.07	2.5	3.09	185,762
Congo	342,000	213,400	0.10	4.7	3.35	179,707
Democratic Republic of the Congo	2,645,000	1,056,500	0.16	12.3	3.25	728,736
Angola	1,247,000	29,000	1.38	6.1	2.85	12,292
Central African Republic	623,000	35,900	0.14	3.7	2.98	24,192
Africa						
Rwanda	26,000	1,010	2.38	196.4	3.41	0
Burundi	28,000	140	2.86	147.7	2.91	0
Uganda	236,000	7,500	1.33	53.4	3.47	436
Tanzania	945,000	14,400	0.69	19.7	3.68	1,100
Kenya	583,000	6,900	1.59	28.6	3.81	44

rces: Data from Barnes 1990; FAO 1993; Chapman et al. 1999. Projected area of forest remaining by 2040 is from Barnes 1990.

with pine or eucalyptus) would meet only 5% of the yearly fuelwood needs of the country.

The reduction in forested area in West Africa and the decline in timber production have encouraged timber companies to increasingly turn their attention from West Africa to Central Africa (Martin 1991). The lessons learned in West Africa have a great deal to offer planners considering various conservation activities in Central Africa. For example, in 1945, the Tropical Shelterwood System (TSS), which cuts vines and poisons unwanted species of medium size, was introduced to manage forest reserves in Nigeria and Ghana; between 1958 and 1970, 188 tonnes of sodium arsenite was used in Ghana to poison trees (Martin 1991). This system was abandoned when it was realized that natural regeneration under this system did not meet expectations (Martin 1991, Chapman and Chapman 1997). In the taungya system, first attempted in Nigeria in 1945, local farmers were allocated land after logging; subsequently they were to plant timber species, tend the trees, then move on. The social consequences of this approach, such as the increased immigration into areas and refusal to relocate, are just now becoming clear (Oates 1995). Such experiments need to be carefully evaluated before planning extraction regimes in Central Africa.

It is clear that with increasing human populations there will be increased demands placed on the remaining forests to meet fuelwood and building needs. Barnes (1990) demonstrated that the most important predictors of the extent of deforestation in tropical Africa were the amount of forest remaining and the size of the human population. For the countries considered in Table 11.1, the average population density is 41 people/km^2 and the human population growth rate is 3.16% per year. Predicting trends in human populations is difficult, but projections suggest that human numbers will continue to increase for another 40 to 50 years (Anderson et al. 1988).

In terms of impacts to aquatic systems, statistics such as these are biased because logging activity and human settlement are frequently associated with river systems (Barros and Uhl 1995; McMillan and Calamari, chapter 21). Rivers are often used to transport logs from the forest from which they are cut to areas closer to markets. For example, timber extracted along the Amazon River can be taken to market by barge at one-third the cost of using truck transport (Barros and Uhl 1995). Further, with the decline in water-borne diseases (see chapter 21), river systems can facilitate movement to areas where road access is difficult and thereby promote settlement.

Africa's Parks and the Protection of Aquatic Ecosystems

Clearly, Africa's forests are under increasing pressure, but one must also consider the extent to which African countries are protecting forests and their associated aquatic systems. The level of protection for forested lands within different regions is limited and constantly changing. Within African countries with closed-canopy forest, an average of 3.2% of each country's area has been protected in national parks or similarly protected areas (IUCN 1985) (Table 11.2). However, the investment of different countries in national parks is always changing, so it is difficult to interpret the significance of the values given in Table 11.2. New parks are being created in some countries, while in other countries parks are being heavily degraded or even degazetted. For example, the northern part of Taï National Park, Ivory Coast (730 km^2), which comprises 21% of the total park area, was temporarily degazetted and is now heavily affected by logging and human agricultural activity (IUCN 1987). Similarly, Bia National Park in Ghana was gazetted in 1974 to include 306 km^2; it was reduced in size to 230 km^2 in 1979 and further reduced to 78 km^2 in 1980. The area excised from the park was classified as a Game Production Reserve, now called a Resource Reserve (IUCN 1987) and has been largely opened up to timber exploitation. The status of the 5,000-km^2 Lopé Reserve of central Gabon is a matter of debate. It is now called Aire d'Exploitation Rationelle de la Faune. With the recent construction of a railroad, it has become more economically feasible to extract timber from the area, and thus logging has intensified (White 1994). However, a recent decree recognized a central core where hunting and logging are banned and a peripheral zone open to logging. In contrast to these examples, the amount of protected land in Uganda had increased from 7,698 km^2 in the 1980s to 11,145 km^2 by 1995 (Table 11.2).

It should be cautioned that values such as those presented in Table 11.2 represent the theoretical maximum protected area. Given the current lack of funding in many regions for enforcement of and education regarding existing wildlife laws, many national parks suffer serious encroachment. For example, during the years of political and economic instability in Uganda, farmers encroached into the Kibale National Park whose area is 766 km^2, and now abandoned farms and degraded forest cover 146 km^2 (Chapman and Lambert 2000). Furthermore, given the current situation of political unrest in many areas, particularly in the Democratic Republic of the Congo (DRC), Rwanda, and Burundi, it is difficult to know the current status of protected areas in those regions (Hart and Hall 1996).

Table 11.2. **Tropical African countries containing closed-canopy forest and description of their park systems, based on IUCN 1985, 1987 (Categories 1 and 2 from the IUCN 1985, 1987)**

Country	Size	Number of national parks	Size of national parks	Percentage of the country protected	Recent changes
Senegal	197,000	6	10,094	5.12	
Guinea-Bissau	36,000	No areas listed	0		
Guinea	246,000	1	130	0.05	
Sierra Leone	72,000	1	980	1.36	A
Liberia	111,000	1	1,307	1.18	
Côte d'Ivoire	322,000	9	17,920	5.57	
Ghana	239,000	6	11,626	4.86	
Togo	57,000	No areas listed	0		
Benin	112,620	2	8,435	7.49	
Nigeria	394,000	1	9,800	2.50	B
Cameroon	475,000	7	12,079	2.54	C
Equatorial Guinea	28,000	No areas listed	0		D
Gabon	265,000	No areas listed	0		
Congo	342,000	1	1,266	0.37	E
Democratic Republic of Congo	2,645,000	8	87,940	3.32	
Angola	1,247,000	1	9,960	0.80	
Cent. African Republic	623,000	3	28,960	4.65	
Rwanda	26,000	2	2,620	10.08	
Burundi	28,000	No areas listed	0		
Uganda	236,000	4	7,698	3.26	F
Tanzania	945,000	10	170,382	18.03	
Kenya	583,000	23	30,509	5.23	

Note: This listing includes all national parks, many of which are not forested. Recent changes involve parks added to a country's parks system after the IUCN 1985, 1987 publications; this listing is not complete. A, proposed but not fully gazetted; B, Cross River 4,320 km^2, Gashaka 6,363 km^2, Chad Basin 2,358 km^2, Oyo 2,550 km^2, Yankari 2,250 km^2; C, Korup 1,259 km^2; D, Parc national de Monte Alen 800 km^2; E, Odzala 2,830 km^2, Nouabalé-Ndoki 3,866 km^2; F, Kibale 766 km^2, Mt. Elgon 1,145 km^2, Bwindi 321 km^2, Semliki 220 km^2, Mgahinga 29 km^2, Rwenzori 996 km^2 (from Howard 1991, USAID 1992).

Rarely has the location of parks been designed to protect aquatic systems. And even when watercourses have entered into the equation, often only the uppermost reaches of the watershed are protected (e.g., the mountain streams in the Rwenzori Mountains). This strategy neglects the often-diverse lower reaches of the rivers. Frequently, rivers running from degraded areas through a park are used to designate the outer boundary of a national park. Similarly, few parks encompass whole lakes, and even those that do may not be protected. In many African countries, there is a tradition to allow the extraction of aquatic resources within park boundaries (see Mugisha, chapter 20). For example, fishing has historically been allowed in most parts of Queen Elizabeth National Park, Uganda, including lakes Edward and George. This practice has led to repeated examples of conflict between park management and the fishing villages within the park (e.g., the taking of fuelwood, hunting). Furthermore, biological data do not exist to evaluate whether harvest levels are sustainable. In countries that do not traditionally allow such extraction, community-based conservation groups, often backed by foreign nongovernmental organizations or donor agencies, are now encouraging parks to allow exploitation of aquatic resources within the parks' boundaries (Wainwright and Wehrmeyer 1998; chapter 20).

Despite the terrestrial bias in park planning and discouraging examples of increased unmonitored exploitation within national parks, there are examples of attention being given to aquatic systems. Four countries share the waters of Lake Tanganyika (DRC, Tanzania, Zambia, and Burundi). It is widely recognized that the management of the lake is a common property and that activities in one part of the lake affect the whole lake (Coulter and Mubamba 1993). There are recent initiatives in all four countries to establish national parks that would not only protect areas of the watershed but would also extend into the lake itself. Similarly, the countries bordering Lake Victoria (Uganda, Kenya, and Tanzania) have obtained World Bank funding to investigate future management options for the lake that would help preserve the lake's biodiversity (see Kaufman, chapter 12).

Clearly, within Africa, tropical forests are increasingly threatened by accelerating rates of forest conversion and degradation (Lanly et al. 1991, FAO 1993). It is also clear that goods and services provided by the aquatic systems connected to these forests are going to become increasingly important to local people. Today, 240 million Africans live where water availability is either only just above or already below a reasonable level to support human habitation (Engelman and LeRoy 1993, Stiassny 1996). Water scarcity will intensify as population densities increase (see Day, chapter 3). Traditionally, conservation efforts have concentrated on the establishment of national parks in pristine or semipristine habitats, but this is only one strategy to protect aquatic communities. The majority of tropical forests and their associated aquatic systems are outside of parks and have experienced or will soon experience significant pressure from activities of humans. It seems inevitable that African countries will turn to timber exploitation as a means of raising income. High levels of foreign debt, with a mean of 58% gross national product for sub-Saharan countries and ranging as high as 241%, place strong pressure on governments to encourage timber harvesting (Stuart et al. 1990). This trend calls for investigations into the long-term effects of different patterns and intensities of forest conversion on downstream aquatic systems. Unfortunately, there are few data on the impacts of forest conversion on aquatic systems in Africa.

A review of the subjects addressed by the contributed articles in the journal *Conservation Biology* between 1994 and 1998 illustrates the paucity of data on conservation issues in African aquatic systems. Between those years, approximately 500 contributed papers were published. Of those, 9% dealt with aquatic systems (marine or fresh water) or animals totally dependent on aquatic systems. Only 1% of the 500 studies dealt with aquatic systems or animals from Africa, and none of those papers dealt with the effects of deforestation or conversion of forest to agricultural land on aquatic systems. This paucity of data limits the ability of the scientific community to evaluate potential impacts of deforestation on aquatic systems.

Impacts of Deforestation on Aquatic Systems

Siltation and Sedimentation

An effect of deforestation readily apparent to even the most casual observer looking at water coming off of recently deforested lands is high siltation and sedimentation. There are many ways to remove logs from a logging site (Putz et al. 2001). 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 the site (e.g., pitsawing) and the timber is hauled out manually, there is relatively little damage to neighboring trees or to the soil (Chapman and Chapman 1996). However, more typically, tractors or bulldozers are used to haul out whole logs. When this method is used, roads must be constructed to allow the machinery access. Logging with bulldozers in Brazil provides a vivid example of the effects of this type of logging. The logging removed only 2% of the trees that were greater than 10 cm in diameter at breast height, but it damaged 26% of the remaining trees and opened the canopy by 50%; in addition, logging roads covered 8% of the forest floor (Uhl and Viera 1989, Frumhoff 1995, Johns et al. 1997). Uprooted trees and logging roads expose the soil to the rains and create situations in which sediment readily flows off the land into adjacent aquatic systems.

The gaps created by tree removal produce higher soil moisture levels because of reduced rainfall interception and uptake by trees. Reduction of rainfall interception and uptake also increase water runoff. L.A. Bruijnzeel (pers. comm.) demonstrated that the felling or poisoning of 40% of upperstory trees corresponds to a 55–70% increase in the water yield. This increase can last for several years, because the regrowth that follows logging has a smaller total leaf area that does not intercept rainfall to the same extent as mature vegetation (Uhl and Jordan 1984). This secondary vegetation has a less developed root system that cannot exploit soil moisture levels to the extent of mature forest trees.

The most detailed tropical studies of how logging affects sediment loads in rivers have been conducted in Malaysia. 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, by cutting vines before felling, by minimizing road construction, and sometimes by cutting cross-drains on steep logging roads. For a dipterocarp forest in peninsular Malaysia, Kasran (1988) demonstrated that sediment yield increased by 97% after conventional logging and by 70% after reduced-impact logging. 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 resulted in only a twofold increase in those characters. At a different study site, Kasran and Nik (1994) found that suspended sediment yield increased 177% in the year after logging and further increased 297% in the following year. Four years after, with the growth of secondary vegetation, the yield recovered to prelogging levels. Reviewing studies in Malaysia, Douglas et al. (1993) documented that logging or ground clearance increased river sediment yields by 2 to 50 times.

These studies in Malaysia are exactly 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 what is happening in Africa. First, studies have shown huge variation in the amounts of sediments eroding from logged areas, depending on the slope, the extent of road construction, and the underlying geology (Douglas et al. 1993). Second, much of the sediment loss from logged areas can be deposited during very brief storm events, illustrating the need for intensive monitoring over several years, a level of monitoring that is not often done. Finally, and possibly most important, the nature of the extraction in Africa and Southeast Asia is different. African forests are very heterogeneous in their species composition. Consequently, harvesting activities produce a comparatively low volume per hectare compared with Southeast Asia. African yields are generally not more than 15 m^3/ha, whereas 60 m^3/ha are commonly extracted from Southeast Asian forests (Bakouma and Buttoud 1996).

Few studies in Africa provide data comparable to those of Malaysian studies. However, in a general survey of watersheds, Dunne (1979) demonstrated that forested catchments in Kenya lose 20–30 tonnes of sediment per square kilometer per year. Sediment yields of agricultural lands vary enormously with runoff. In the wettest, steepest cultivated catchments, soil loss exceeds 4,000 tonnes/km^2 per year. The sediment yields from rangeland catchments are also variable; the driest catchments lose less than 100 tonnes/km^2 per year, whereas up to 20,000 tonnes/km^2 per year are exported from the wettest, steepest grazed catchments (Dunne 1979).

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 et al. 1996, Richards et al. 1996, Stone and Wallace 1998). Human modifications to the landscape, such as changing the amount of silt in the water or changing the light level, will affect the aquatic biota.

These changes are complex and difficult to predict, and there are too few studies available from tropical areas to make generalizations that are sufficiently reliable to use in management. In Oregon, Carlson et al. (1990) documented that macroinvertebrate density was 20–113% greater at logged sites than at unlogged sites, despite similar diversity between the logged and unlogged sites. In contrast, sand deposition resulting from stream diversion in Cornwall, United Kingdom, was associated with a dramatic reduction in aquatic plants, the elimination of several species, and a lower diversity of macroinvertebrates (Nuttall 1972). In one of the longest-term studies conducted to date, Stone and Wallace found that, 16 years after logging, benthic invertebrate abundance was three times higher in the disturbed stream than in undisturbed streams in the same watershed.

Changes in vegetation and macroinvertebrate abundance and diversity will alter food availability for fishes. A study conducted in Maine documented that, after a decline in aquatic prey resources resulting from clear-cut logging, one fish species declined in abundance while a second species increased in abundance because it switched to eating terrestrial arthropods (Garman and Moring 1993). In a logged eucalyptus forest in Tasmania, Davies and Nelson (1994) found that a significant decrease in the absolute abundance of riffle macroinvertebrates, particularly stoneflies (Plecoptera) and mayflies (Emphemeroptera), corresponded to a decrease in brown trout (*Salmo trutta*) abundance. Fish populations will also be affected when sediment fills the interstitial spaces between cobbles in riffle areas and smothers eggs and eliminates habitat for fry or when it fills pools (Waters 1995, Pringle and Benstead 2001). If sufficient sediment fills riverbeds or streams it can change the structure of the invertebrate community because invertebrate populations are closely tied to the particle size of streambed sediments (Waters 1995). Heavy sediment loads will embed rocks and cobbles in sediment and reduce habitat suitable for some major insect groups such as Ephemeroptera, Plecoptera, and Trichoptera. These groups will be replaced by small burrowing forms such as Chironomidae and oligochaetes. Such burrowing forms may be less available to foraging fishes (Pringle and Benstead 2001).

Cohen et al. (1993) studied the effects of sedimentation on the species richness of three taxonomic groups in Lake Tanganyika (ostracods, fish, and diatoms) 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 1991, Cohen et al. 1993). In the north end of the lake, where deforestation approaches 100% of the forested land (Cohen et al. 1993), rates of soil erosion range between 28 and 100 tonnes/ha per year, depending on the slope (Bizimana and Duchafour 1991). The species richness of ostracods and fish appeared to be heavily affected by sediment, but diatoms showed little change in species richness (Cohen et al. 1993).

Effects of Removing Overhanging Trees

When one thinks of riverine systems in Africa, one typically thinks of rivers like the Congo, Nile, Niger, and Zambezi. However, in general, Africa is drier than other tropical continents, and thus more than 90% of Africa's river length is composed of rivers less than 9.0 km long (Stiassny 1996). As a result, the majority of African rivers 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), and increases light penetration and water temperature. Increased light resulting from the removal or loss of the canopy leads to higher standing stocks of algae (Ulrich et al. 1993). Such changes can restructure the macroinvertebrate communities. For example, Wallace and Gurtz (1986) documented that mayfly (Ephemeroptera) production was 28 times higher in a logged catchment than in an unlogged stream. Because the mayfly is a grazing invertebrate, this difference was presumed to have been the result of an increase in primary productivity associated with the opening of the canopy and increased light levels.

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 communities of logged streams, a stream with a 100-m buffer zone, and control streams. They found that macroinvertebrate community structure in the buffered stream was different from that in the undisturbed streams, but it was more similar to that of the undisturbed streams than to that of the logged streams. These results suggest that a 100-m-wide buffer zone can effectively ameliorate disturbance due to clear-felling (see also Richards et al. 1996). Another study (Davies and Nelson 1994) documented the effects of both cable and conventional logging in areas with a range of riparian buffer strip widths (0–50 m). They found that all impacts were significant only at buffer widths of less than 30 m.

Recommendations

Human modification to forests has a variety of impacts on associated aquatic systems. These impacts depend on the type and intensity of modification, subsequent land treatment, the original aquatic community, the underlying soil type and geology, the amount and pattern of rainfall in the region, and a variety of other factors. To date, there is little information available on effects of deforestation or on different management options to mitigate potential effects. However, current estimates of forest loss and projected rates of deforestation in Africa require that management decisions be made now. These decisions must be based on what is currently known, and planning must be undertaken to most efficiently use research efforts. From this perspective the following recommendations are proposed to channel conservation and research efforts in the most appropriate fashion.

The paucity of African studies cited in this chapter clearly illustrates that more research is needed to determine the impacts that various forms of human modification to terrestrial habitats will have on African aquatic systems. Particularly pressing issues include (1) the level of sediment loading that occurs after different types of land-use change, (2) the effect of increased silt and sediment loads on aquatic systems, (3) changes in plant and animal communities associated with canopy removal above rivers, and (4) cascading effects of altering food webs.

Given high levels of foreign debt and past reliance on aquatic systems to provide resources, it is clear that aquatic resources will continue to be exploited. The exploitation of terrestrial and aquatic systems is often assessed using naive economic criteria, and more detailed assessments of the costs of different management options are needed. For example, the actual costs of land uses, crop production, and other economic activities in watersheds need to be determined so that gains occurring on terrestrial systems can be assessed in terms of losses to aquatic systems.

Studies are needed that document how aquatic disturbances are mitigated by different management techniques. Mitigating techniques include use of (1) riparian buffer strips in logged, agricultural, and pastoral lands, (2) barriers to reduce roadside runoff, (3) reduced-impact logging, and (4) terracing of agricultural land.

Protection and conservation of aquatic systems in Africa require enactment of policy changes. All too often, there is a firm scientific understanding of the need to change the way in which people are exploiting resources, but this understanding is not put into action because policies are not changed or enforced. Policy changes that should be evaluated include (1) establishing parks or reserves with a consideration of how to protect aquatic systems and (2) recognizing that modifications to terrestrial systems will often result in detrimental modification to aquatic systems.

Sometimes, there are sufficient scientific data to understand how resource extraction should be modified and there is policy in place to specify how the extraction process should be done, but no change in practice occurs. Typically in such cases, the local people are not convinced of the need for change or are insufficiently motivated to modify their practices. Thus, local people must be involved in decision making and the implementation of change.

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Bibliography

Anderson, R.M., R.M. May, and A.R. Maclean. 1988. Possible demographic consequences of AIDS in developing countries. *Nature* 332: 228–234.

Bakouma, J., and G. Buttoud. 1996. African markets: A future for African sawnwood? *Tropical Forest Update* 6: 17.

Barnes, R.F.W. 1990. Deforestation trends in tropical Africa. *African Journal of Ecology* 28: 161–173.

Barros, A.C., and C. Uhl. 1995. Logging along the Amazon River and estuary: Patterns, problems and potential. *Forest Ecology and Management* 77: 87–105.

Bizimana, M., and H. Duchafour. 1991. A drainage basin management study: The case of the Ntihangwa River basin. Pages 43–45 in *Report of the first international conference on conservation and biodiversity of Lake Tanganyika*, ed. A. Cohen. Washington, DC: Biodiversity Support Program.

Boahene, K. 1998. The challenge of deforestation in tropical Africa: Reflections on its principal causes, consequences and solutions. *Land Degradation and Development*. 9: 247–258.

Carlson, J.Y., C.W. Andrus, and H.A. Froelich. 1990. Woody debris, channel features, and macroinvertebrates of streams with logged and undisturbed riparian timber in northeastern Oregon, U.S.A. *Canadian Journal of Fisheries and Aquatic Science* 47: 1103–1111.

Chapman, C.A., and L.J. Chapman. 1996. Exotic tree plantation and the regeneration of natural forests in Kibale National Park, Uganda. *Biological Conservation* 76: 253–257.

Chapman, C.A., and L.J. Chapman. 1997. Forest regeneration in logged and unlogged forests of Kibale National Park, Uganda. *Biotropica* 29: 396–412.

Chapman, C.A., and J.E. Lambert. 2000. Habitat alteration and conservation of African primates: A case study of Kibale National Park, Uganda. *American Journal of Primatology* 50: 169–186.

Chapman, C.A., A. Gautier-Hion, J.F. Oates, and D.A. Onderdonk. 1999. African primate communities: Determinants of structure and threats to survival. Pages 1–37 in *Primate communities,* ed. J.G. Fleagle, C.H. Janson, and K. Reed. Cambridge: Cambridge University Press.

Cohen, A., ed. 1991. *Report of the first international conference on conservation and biodiversity of Lake Tanganyika.* Washington, DC: Biodiversity Support Program.

Cohen, A., R. Bills, C.Z. Cocquyt, and A.G. Caljon. 1993. The impact of sediment pollution on biodiversity in Lake Tanganyika. *Conservation Biology* 7: 667–677.

Coulter, G.W., and R. Mubamba. 1993. Conservation in Lake Tanganyika, with special reference to underwater parks. *Conservation Biology* 7: 678–685.

Davis, P.E., and M. Nelson. 1994. Relationships between riparian buffer widths and the effects of logging on stream habitat, invertebrate community composition and fish abundance. *Australian Journal of Marine and Freshwater Research* 45: 1289–1305.

Douglas, I., T. Greer, K. Bidin, and M. Silsbury. 1993. Impacts of rainforest logging on river systems and communities in Malaysia and Kalimantan. *Global Ecology and Biogeography Letters* 3: 245–252.

Dunne, T. 1979. Sediment yield and land use in tropical catchments. *Journal of Hydrology* 42: 281–300.

Engelman, R., and P. LeRoy. 1993. *Sustaining water, population and the future of renewable water supplies.* Washington, DC: Population Action International.

FAO. 1993. *Forest resource assessment, 1990: Tropical countries.* FAO Forestry Paper No. 112. Rome: FAO.

Fore, L.S., J.R. Karr, and R.W. Wisseman. 1996. Assessing invertebrate responses to human activities: Evaluating alternative approaches. *Journal of the North American Benthological Society* 15: 212–231.

Frumhoff, P.C. 1995. Conserving wildlife in tropical forests managed for timber. *Bioscience* 45: 456–464.

Garman, G.C., and J.R. Moring. 1993. Diet and annual production of 2 boreal river fishes following clear-cut logging. *Environmental Biology of Fishes* 36: 301–311.

Growns, I.O., and J.A. Davis. 1991. Comparison of the macroinvertebrate communities in streams in logged and undisturbed catchments 8 years after harvesting. *Australian Journal of Marine and Freshwater Research* 42: 689–706.

Hart, J.A., and J.S. Hall. 1996. Status of eastern Zaire's forest parks and reserves. *Conservation Biology* 10: 316–327.

Howard, P.C. 1991. *Nature conservation in Uganda's tropical forest reserves.* Gland, Switzerland: IUCN.

IUCN. 1985. *United Nations list of national parks and protected areas.* Gland, Switzerland: IUCN.

IUCN. 1987. *IUCN directory of afrotropical protected areas.* Gland, Switzerland: IUCN.

Johns, J.S., P. Barreto, and C. Uhl. 1997. Logging damage in planned and unplanned logging operations in the eastern Amazon. *Forest Ecology and Management* 89: 59–77.

Kasran, B. 1988. Effect of logging on sediment yield in a hill dipterocarp forest in Peninsular Malaysia. *Journal of Tropical Forest Science* 1: 56–66.

Kasran, B., and A.R. Nik. 1994. Suspended sediment yield resulting from selective logging practices in a small watershed in Peninsular Malaysia. *Journal of Tropical Forest Science* 7: 286–295.

Kaufman, L.S. 1992. Catastrophic change in species rich freshwater ecosystems: the lessons of Lake Victoria. *Bioscience* 42: 846–858.

Kaufman, L.S., L.J. Chapman, and C.A. Chapman. 1997. Evolution in fast forward: Haplochromine fishes of the Lake Victoria region. *Endeavour* 21: 23–30.

Lanly, J.P., K.D. Singh, and K. Janz. 1991. FAO's 1990 reassessment of tropical forest cover. *Nature and Resources* 27: 21–26.

Martin, C. 1991. *The rainforests of West Africa: Ecology, threats, and conservation.* Basel: Borkhauser Verlag.

Mayaux, P., F. Achard, and J.P. Malingreau. 1998. Global forest area measurements derived from coarse resolution satellite imagery: A comparison with other approaches. *Environmental Conservation* 25: 37–52.

Mohd, Z.-I. 1994. Zoogeography and biodiversity of the freshwater fishes of Southeast Asia. *Hydrobiologia* 285: 41–48.

Nuttall, P.M. 1972. The effects of sand deposition upon the macroinvertebrate fauna of the River Camel, Cornwall. *Freshwater Biology* 2: 181–186.

Oates, J.F. 1995. The dangers of conservation by rural development—A case study from the forests of Nigeria. *Oryx* 29: 115–122.

Pringle, C.M., and J.P. Benstead. 2001. Effects of logging on tropical river ecosystems. Pages 305–325 in *Conserving wildlife in managed tropical forests,* ed. A. Grajal, J. Robinson, and R. Fimbel. New York: Columbia University Press.

Putz, F.E., L.K. Sirot, and M.A. Pinard. 2001. Tropical forest management and wildlife: Sivicultural effects on forest structure, fruit production, and locomotion of non-volant arboreal animals. In *Conserving wildlife in managed tropi-*

cal forests, ed. A. Grajal, J. Robinson, and R. Fimbel. New York: Columbia University Press.

Richards, C., L.B. Johnson, and G.E. Host. 1996. Landscape-scale influences on stream habitats and biota. *Canadian Journal of Fisheries and Aquatic Sciences* 53: 295–311.

Stiassny, M.L.J. 1996. An overview of freshwater biodiversity: With some lessons from African fishes. *Fisheries* 21: 7–13.

Stone, M.K., and J.B. Wallace. 1998. Long-term recovery of a mountain stream from clearcut logging: The effects of forest succession on benthic invertebrate community structure. *Freshwater Biology* 39: 151–169.

Struhsaker, T.T. 1981. Forest and primate conservation in East Africa. *African Journal of Ecology* 19: 99–114.

Stuart, S.N., R.J. Adams, and M.D. Jenkins. 1990. *Biodiversity in sub-Saharan Africa and its island: Conservation, management, and sustainable use.* Gland, Switzerland: IUCN.

Uhl, C., and C.F. Jordan. 1984. Succession and nutrient dynamics following forest cutting and burning in Amazonia. *Ecology* 65: 1476–1490.

Uhl, C., and I.C.G. Viera. 1989. Ecological impacts of selective logging in the Brazilian Amazon: A case study from the Paragominas region of the State of Pará. *Biotropica* 21: 98–106.

Ulrich, K.E., T.M. Burton, and M.P. Oemke. 1993. Effects of whole-tree harvest on epilithic algal communities in headwater streams. *Journal of Freshwater Ecology* 8: 83–92.

Wainwright, C., and W. Wehrmeyer. 1998. Success in integrating conservation and development? A study from Zambia. *World Development* 26: 933–944.

Wallace, J.B., and M.E. Gurtz. 1986. Response of *Baetis* mayflies (Ephemeroptera) to catchment logging. *American Midland Naturalist* 115: 25–41.

Waters, T.F. 1995. *Sediment in streams: Sources, biological effects, and control.* American Fisheries Society Monograph 7. Bethesda, MD: American Fisheries Society.

White, L.J.T. 1994. The effects of commercial mechanized selective logging on a transect in lowland rainforest in the Lopé Reserve, Gabon. *Journal of Tropical Ecology* 10: 313–322.

Yusop, Z., and A. Suki. 1994. Effects of selective logging methods on suspended solids concentration and turbidity level in streamwater. *Journal of Tropical Forest Science.* 7: 199–219.

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