

# Rapid Land-Use Change and Its Impacts on Tropical Biodiversity

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Rates of forest conversion are extremely high in most tropical regions and these changes are known to have important impacts on biotas and ecosystems. I summarize available information on responses of wildlife and plant communities to habitat fragmentation, selective logging, surface fires, and hunting, which are four of the most widespread types of tropical land-use change. These changes alter forest ecosystems in complex ways and have varying impacts on different animal and plant species. In most human-dominated landscapes, forests are subjected to not one change but to two or more simultaneous alterations, the effects of which can be particularly destructive to tropical biotas. I illustrate this concept by describing the synergistic interactions between habitat fragmentation and surface fires, and between logging, fires, and hunting.

## INTRODUCTION

In this chapter, I provide an overview of the major impacts of land-use change on wildlife and plant communities, especially in the tropics. My focus is on four major changes—habitat fragmentation, selective logging, surface fires, and hunting—that constitute the most immediate threats to tropical biota today [Laurance and Peres, in press, 2004]. Each of these topics has its own technical literature, and I provide strategic references throughout for those who wish to delve further into any particular subject.

In reality, biota in most human-altered landscapes are affected not by a single threat but by several simultaneous perturbations. In the final section of this chapter, I highlight some important synergisms that can arise when multiple threats are operating in concert. It is, however, beyond the scope of this chapter to address the impacts of large-scale atmospheric

and climatic changes on tropical biota [e.g., Pounds *et al.*, 1999; Laurance *et al.*, 2004; Malhi and Phillips, 2004], despite the fact that such phenomena will certainly interact with local land-use changes, posing potentially serious threats to biota in some regions [Williams *et al.*, 2003; Thomas *et al.*, 2004]. It is also not possible to describe the myriad ways in which basic ecological processes, such as pollination, seed dispersal, herbivory, and nutrient cycling, are changed by habitat alteration [e.g., Renner, 1998; W. F. Laurance, in press, 2004]. My focus, rather, is on the ecological impacts of rapid land-use change on animal and plant communities.

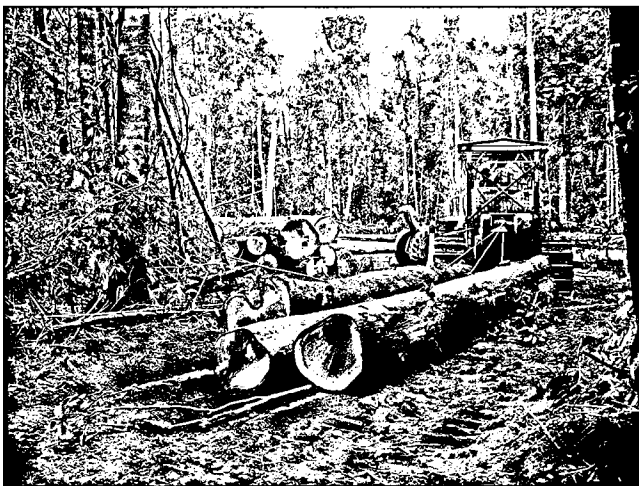
## SELECTIVE LOGGING

Industrial logging is an important and growing land use in the tropics. Each year, roughly 6 million ha of forest is selectively logged—an area twice the size of Belgium [Whitmore, 1997]. Logging has been most intensive in Southeast Asia [Dinerstein and Wikramanayake, 1993; Primack and Lovejoy, 1995; Curran *et al.*, 1999, 2004] and is expanding rapidly in tropical Africa, Central America, the Amazon, and Melanesia (New Guinea and the Solomon Islands) [Laurance, 1998,

1999, 2001a; *Nepstad et al.*, 1999; *W. F. Laurance et al.*, in press, 2004].

Logging in the tropics is nearly always selective. This is because many tree species produce low-quality wood (having hollows or poor wood properties) or grow slowly, or because commercial markets for those species have not been developed. Loggers use bulldozers and other heavy equipment to remove timber (Figure 1). Although commercial loggers typically harvest just 1–30 trees ha<sup>-1</sup>, from 10–40% of the remaining forest biomass is killed or damaged from the labyrinths of roads, bulldozer trails, and small clearings in the forest. This can cause substantial mortality of smaller trees, soil damage and compaction, microclimatic alterations, and sharply increased soil erosion and stream sedimentation [*Uhl and Vieira*, 1989; *Verissimo et al.*, 1992]. Most selective logging regimes are polycyclic, in that forests will be re-logged at nominal intervals of 20–40 years [*Johns*, 1997; *Grajal et al.*, 2001]. Thus, tropical logging is a recurring disturbance.

Many wildlife species decline in abundance in logged forests. Among the most vulnerable groups are many understory insectivorous birds, including terrestrial species, members of mixed flocks, and solitary sallying species [*Driscoll*, 1986; *Thiollay*, 1992; *Plumptre*, 2001], certain primates [*Johns*, 1986; *White and Tutin*, 2001] and arboreal marsupials [*Laurance and Laurance*, 1996], microchiropteran bats adapted for flying in forest interiors [*Crome and Richards*, 1988], and disturbance-sensitive frogs and reptiles [*Preen*, 1981]. Some faunal groups, including rodents [*Malcolm and Ray*, 2000], butterflies [*Willott et al.*, 2000; *Owen*, 2001], and certain lemurs [*Ganzhorn*, 1995], generally increase in abundance or diversity in logged forests because of increased ground cover, foliage, or fruit abundance.



**Figure 1.** Bulldozers and other equipment can cause heavy disturbances in logged tropical forests (photo by W. F. Laurance).

Plant communities and forest architecture also change in response to logging disturbance. Along skid tracks and logging roads, the abundance and diversity of native understory herbs decline, whereas invading herbs from non-rainforest habitats increase sharply in abundance [*Costa and Magnusson*, 2002]. Logging increases the density of light-loving tree species at the expense of slower-growing, shade-adapted species, but evidently does not lead to a marked reduction in tree diversity, at least during the first 1–2 logging cycles [*Nicholson et al.*, 1988; *Cannon et al.*, 1998]. However, commercially valuable tree species, such as mahogany (*Swietenia* spp.), may be locally extirpated by logging [*Fearnside*, 1997]. Moreover, lianas (climbing woody vines that are important structural parasites of trees) thrive in logged forests [*Putz*, 1991; *Pinard and Putz*, 1994] and can suppress tree regeneration and growth [*Fox*, 1968; *Putz*, 1984; 1991], increase tree mortality [*Appanah and Putz*, 1984], and reduce forest biomass [*Laurance et al.*, 2001b].

The ecological impacts of logging, and the time required for disturbance-sensitive species to recover in logged forests, are both positively correlated with harvest intensity [*Johns*, 1997]. In general, harvests are most intensive in Southeast Asian forests, because of the predominance there of trees in the family Dipterocarpaceae (especially in the genus *Shorea*), which are prized timbers. Harvest methods that reduce the ecological effects of selective logging have been developed, but have been applied in only a tiny fraction of all tropical forests [*Bowles et al.*, 1998]. In most tropical countries, damage from selective logging is excessive because enforcement of regulations designed to reduce logging impacts is poor or nonexistent [*Johns*, 1997; *Laurance*, 2000a; *Putz et al.*, 2000].

Although logging can cause substantial changes in faunal and plant communities and a decline of disturbance-sensitive species, there is at present little compelling evidence that logging *per se* leads to significant extinctions of local populations [*Johns*, 1997; *Grajal et al.*, 2001]. This conclusion, however, must be tempered with three important caveats. First, most forests in the tropics have been logged only once or twice, and thus the long-term effects of repeated polycyclic logging are unknown. Second, as discussed below, logging increases forest desiccation and fuel loads, making forests much more vulnerable to destructive wildfires. Finally, in some regions, the labyrinths of forest roads constructed by loggers sharply increase forest incursions by hunters, miners, ranchers, and slash-and-burn farmers, whose activities are often far more destructive to forests than is logging itself.

## FOREST FRAGMENTATION

Habitat fragmentation is a ubiquitous consequence of land-use change. In the tropics, slash-and-burn farming by small-

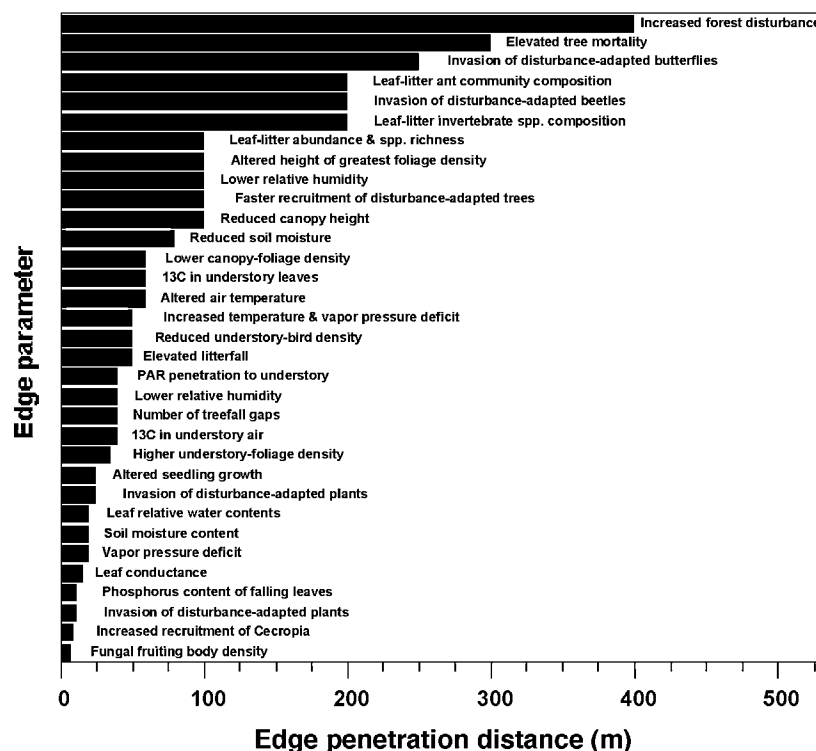
scale landowners is probably the greatest direct cause of forest loss and fragmentation [Myers, 1999], although cattle ranching is also of major importance in Latin America [Laurance, 1999]. In recent decades, the largest remaining tracts of tropical forest in the Amazon [Skole and Tucker, 1993; Laurance et al., 2001a], Southeast Asia [Primack and Lovejoy, 1995; Curran et al., 2004], and tropical Africa [Webber et al., 2001; Laurance et al., in press, 2004] have been markedly reduced and fragmented.

Fauna and plant communities in fragmented landscapes are subjected to a diversity of ecological changes. As deforestation proceeds, some species that are initially present in a landscape may be completely extirpated by habitat loss, and others may persist in tiny numbers in fragments, especially if they are rare or patchily distributed [Wilcox, 1980]. Populations of many forest-dependent species are completely isolated in fragments, whereas others are only partially isolated, depending on their tolerance of degraded habitats and vagility [Diamond et al., 1987; Laurance, 1991]. The degree to which populations in fragments are connected demographically and genetically is also influenced by the distance to which the fragment is isolated from other forest tracts, and the types of degraded land surrounding the fragment. When isolated, small populations (<100 individuals) are vulnerable to random demo-

graphic and genetic effects, frequently exacerbated by environmental variations and local catastrophes [Leigh, 1981; Shafer, 1981]. Collectively, these four phenomena can be a powerful driving force of local extinction.

In most human-dominated landscapes, the large majority of forest fragments are small, ranging from <1 ha to a few hundred ha in area [Gascon et al., 2000; Peres, 2001; Cochrane and Laurance, 2002]. Rates of local extinction are usually high in such fragments, which sustain small population sizes of many species. Small fragments are also vulnerable to edge effects (Figure 2)[AC2]—diverse physical and biotic alterations associated with the abrupt, artificial margins of habitat fragments [Lovejoy et al., 1986; Murcia, 1995; Laurance et al., 2002]. Fragmented tropical forests often have elevated tree mortality, a proliferation of disturbance-adapted trees and vines [Viana et al., 1997; Laurance et al., 1998a, 1998b], and hotter, drier conditions near forest boundaries [Kapos, 1989]. Most edge effects penetrate anywhere from a few meters to a few hundred meters into fragments (Figure 2), although some phenomena, such as surface fires (see below) and some invading species, can penetrate several kilometers into forests [Laurance, 2000b].

Numerous species decline or disappear in tropical forest fragments (<100 ha in area). In the Amazon, this includes



**Figure 2.** Diverse edge effects occur near the abrupt, artificial margins of forest fragments, as shown here for a long-term study in Amazonia [after Laurance et al., 2002].

many species of large mammals [Lovejoy *et al.*, 1986], primates [Gilbert and Setz, 2001], understory birds [Stouffer and Bierregaard, 1995a; Stratford and Stouffer, 1999], and even certain beetle, ant, bee, termite, and butterfly species [Powell and Powell, 1987; Vasconcelos, 1988; Klein, 1989; Souza and Brown, 1994; Brown and Hutchings, 1997; Didham, 1997]. Many species that decline in fragments are sensitive to edge effects, including numerous flies, bees, wasps [Fowler *et al.*, 1993], beetles [Didham, 1997], ants [Carvalho and Vasconcelos, 1999], and butterflies [Brown and Hutchings, 1997]. A number of insectivorous understory birds avoid edges, particularly solitary species, obligatory ant followers, and those that forage in mixed-species flocks [S. G. Laurance, in press, 2004; S. G. Laurance *et al.*, 2004]. In general, the species most vulnerable to fragmentation are those that have large area requirements, are strongly dependent on primary forest, are sensitive to edge effects, and/or are susceptible to hunting (see below).

Some species increase in fragmented forests, sometimes sharply. Certain termites, leafhoppers, scale insects, aphids, and aphid-tending ants [Fowler *et al.*, 1993] increase near edges, probably because foliage and dead trees are more abundant near edges. Light-loving butterflies also increase near edges [Brown and Hutchings, 1997]. Birds that forage in treefall gaps, such as some arboreal insectivores, hummingbirds, and habitat generalists, often become abundant near edges [Stouffer and Bierregaard, 1995a, 1995b]. Frugivorous bats increase near edges, probably because such areas have higher fruit abundance than forest interiors [Kalko, 1998]. The insectivorous marsupial *Metachirus nudicaudatus* apparently increases in fragments because dead trees and ground cover, which provide favored foraging microhabitats, increase near edges [Malcolm, 1997].

The net effect is that fragmentation markedly changes forest ecology, especially in the small (<500 ha) fragments that predominate in most human-dominated landscapes, accelerating forest dynamics, reducing species diversity, and sometimes greatly altering the abundances of species [Laurance *et al.*, 2002]. Fragmented forests are far more vulnerable to species invasions and fires (see below) than are intact forests. In addition, fundamental ecological processes, such as nutrient cycling, carbon storage, pollination, and seed dispersal, can be altered in fragmented forests [W. F. Laurance, in press, 2004].

### SURFACE FIRES

Each year, hundreds of thousands of fires are lit in the humid tropics, to raze forests for slash-and-burn farming and ranching. Although highly destructive to natural ecosystems, these intentional fires are now being rivaled in terms of their

ecological impacts by a more subtle hazard: accidental surface fires [Cochrane, 2003].

Surface fires are deceptively unimpressive, creeping along the forest floor as a thin ribbon of flames and rarely exceeding 20 cm in height (Figure 3). [AC3] Unlike fast-burning intentional fires, which are lit after slashing and felling vegetation to dry out forests and increase flammability, surface fires merely consume the leaf and woody litter and generate only modest heat, covering as little as 150 m in a day [Cochrane and Schulze, 1999].

Nevertheless, surface fires kill many rainforest plants, which typically have thin (<1 cm-thick) bark and thus are vulnerable to heat stress [Uhl and Kauffman, 1990; Barlow *et al.*, 2003a]. Studies conducted 1–2 years after a surface fire suggest that the slow-moving flames can girdle and kill a third or more of all trees (at least 10 cm in diameter) and most vines



**Figure 3.** A destructive surface fire creeping along the forest floor in eastern Amazonia (photo by M. A. Cochrane).

and ground forbs [Cochrane and Schulze, 1999; Cochrane et al., 1999; Nepstad et al., 1999; Barlow et al., 2003b]. Saplings and small trees, which have thinner bark than large trees, are especially vulnerable.

Recent studies in central Amazonia reveal that surface fires have serious effects on forest wildlife. One large fire (1100 km<sup>2</sup>), which occurred during the strong El Niño drought of 1997–98, killed or injured numerous animals, including monkeys, marmosets, sloths, peccaries, deer, parrots, toucans, snakes, and lizards. Most vulnerable were species with low mobility, poor climbing ability, or a reliance on cavity nests in trees [Barlow et al., 2002; Peres et al., 2003]. Burnt forests were nearly silent, even at dawn and dusk when the vocal activity of birds and monkeys peak, suggesting that even mid-story and canopy species had been killed or driven away by smoke asphyxiation, or fled when maturing fruit crops in damaged trees were aborted. The initial declines of wildlife were exacerbated by local hunters, who killed many stressed or injured animals [Peres et al., 2003].

By 10–15 months after the fire, an average of 36% of all trees, three-quarters of all saplings, and many lianas had died. Canopy openness was four times greater than in unburned forest (ranging from 12–32%) and light levels had risen dramatically in the forest understory [Barlow et al., 2003b; Peres et al., 2003]. As a result, a dense flush of vegetation, dominated by disturbance-loving pioneer trees, bamboo, sedges, and coarse herbs, had proliferated in the understory.

These changes in forest structure and microclimate had major effects on bird communities [Barlow et al., 2002]. Mist-net samples collected one year after the fire revealed that understory birds were much less abundant in burnt than unburnt forest, and the trophic structure of bird communities was altered. Specialized insectivores such as army ant-followers, dead-leaf gleaners, terrestrial gleaners, and arboreal sallying insectivores were most severely affected, whereas nectar- and seed-eating species increased markedly. The birds most affected by the fire were uncommon, disturbance-sensitive, and habitat specialists—the same species that are highly vulnerable to habitat fragmentation and logging [Barlow et al., 2002; Peres et al., 2003; Barlow and Peres, 2004].

Of even greater concern, surface fires can initiate an irreversible process of forest degradation. The initial fire in rainforests typically kills 10–40% of all trees and most vines [Barbosa and Fearnside, 1999; Cochrane et al., 1999; Barlow et al., 2003b]. As a result, the forest canopy becomes fragmented and the quantity of dead fuels rises as dead leaves and trees begin to fall. Soon, the forest is far more prone to subsequent fires, because the diminished canopy allows rapid drying and the dying vegetation provides large quantities of combustible fuel [Cochrane and Schulze, 1999; Cochrane et al., 1999].

The second fire is usually much hotter and more destructive than the initial burn, killing 40–50% of remaining trees [Cochrane et al., 1999] and overwhelming even large, thick-barked trees [Peres, 1999; Barlow et al., 2003b]. Moreover, whereas the initial surface fire might require an extensive drought, subsequent fires can occur after just a few weeks without rain. During the first several fires, more fuels are created than destroyed, and a positive feedback results in which each fire becomes more likely and intensive. This process can eradicate rainforest trees and promote extensive grass invasion, converting rainforests into anthropogenic savanna or degraded scrub. In regions with strong dry seasons, once initiated this degradation process may be nearly irreversible [Cochrane and Schulze, 1999].

## HUNTING

Over-hunting is one of the most pervasive threats to tropical wildlife and has grown sharply in intensity, for three reasons. First, human populations in the tropics have more than quadrupled in the last century, and affluence has also risen in many areas, sharply increasing demand for wild meat [Ehrlich et al., 1995; Laurance, 1999; Myers, 1999]. Second, hunting technologies have improved significantly; traditional weapons like bows, spears, and nets have now been almost completely replaced by shotguns and cable snares, which are far more effective [Alvard et al., 1997; Noss, 1998; Milner-Gulland et al., 2003]. Finally, refuges for forest wildlife have declined because the total area of forest has fallen sharply and because many remaining forest tracts are penetrated by roads, and thus are more accessible to hunters [Peres and Terborgh, 1995; Laurance, 1998; Wilkie et al., 2000; Curran et al., 2004].

Over-hunting alters the structure of animal communities. Especially vulnerable are large-bodied mammals, such as antelope, deer, tapirs, elephants, bushpigs, peccaries, larger primates, large rodents, and top carnivores [Robinson and Redford, 1991; Peres, 2000], species with low reproductive rates [Bodmer et al., 1997], and other large or conspicuous vertebrates [Fa et al., 2000; Milner-Gulland et al., 2003]. These changes can have important impacts on forest ecosystems. The extirpation of large predators, for example, may lead to population explosions of seed predators and generalist omnivores that have important impacts on nesting birds and large-seeded tree species [Terborgh, 1992]. Hunting-related declines of seed predators and dispersers can disrupt normal plant-recruitment processes and may cause a decline in local tree diversity [Chapman and Chapman, 1996; Dirzo, 2001].

Chronic over-hunting has become the norm in many tropical areas. Exploited species often decline sharply in abundance and biomass within hiking distance (10–15 km) of

villages and roads [Barnes *et al.*, 1991; Peres, 2000; Wilkie *et al.*, 2000; Lahm, 2001]. In addition, commercial hunters use hunting camps and logging tracks to penetrate deep into remaining forest tracts [Wilkie *et al.*, 2000; Lahm, 2001]. Even many nature reserves are subjected to heavy hunting pressure [Peres and Terborgh, 1995; Lahm, 2001; Oluput and Chapman, in press, 2004], which can increase local extinctions of wildlife [Woodroffe and Ginsberg, 1998].

Hunting pressure varies among the major tropical regions. In Brazilian Amazonia, as many as 23.5 million vertebrates, equivalent to about 90,000 tons of wild meat, are consumed annually by rural populations [Peres, 2000]. Hunting pressure is probably even higher in Southeast Asia because of the proliferation of logging roads and forest colonization [Bennett and Gumal, 2001]. The most intensively hunted region, however, is tropical Africa. The estimated total harvest of wild mammals in the Congo Basin approaches 5 million tons annually, which is roughly 30 times higher than that in Neotropical forests [Fa *et al.*, 2002]. Many species are seriously overexploited; for example, of 57 hunted vertebrate species in the Congo Basin, 60% are hunted unsustainably [Fa *et al.*, 2002]. In West Africa, acute over-hunting has led to a near-collapse of the commercial bushmeat trade and the extinction of a primate species [Oates *et al.*, 2000].

#### SYNERGISTIC EFFECTS OF MULTIPLE THREATS

Here I have surveyed four of the most important anthropogenic threats to tropical ecosystems. In most landscapes, however, wildlife and plant communities are rarely affected by a single environmental change but by several changes operating simultaneously. As I will show here, the synergistic effects of multiple threats can sharply increase their impacts on tropical biotas.

##### *Logging and Its Synergisms*

Industrial logging causes moderate to heavy forest disturbance (depending on logging intensity), but most wildlife species can persist in logged forests, albeit at reduced densities [Johns, 1997]. Of greater importance is that, by creating labyrinths of forest roads, logging can facilitate invasions of forests by hunters, miners, ranchers, and slash-and-burn farmers, which can severely degrade or destroy forests [Laurance, 2001b]. In frontier areas of Borneo, New Guinea, the Amazon, and central Africa, I have observed slash-and-burn farmers following virtually in the footsteps of loggers. The mosaics of farming plots and scrubby regrowth they produce bear little resemblance to primary forest, and are of limited value for wildlife conservation.

Logging also typically leads to a marked rise in commercial and subsistence hunting, by providing hunters with greater access to unexploited wildlife populations and lowering the cost of transporting wild meat to market [Wilkie *et al.*, 1992, 2000]. The loggers themselves are often avid hunters, with bushmeat sales providing up to 40% of their annual income [Wilkie *et al.*, 2001]. Loggers also rely on wildlife for food; for instance, employees at individual logging camps in Malaysia [Bennett and Gumal, 2001] and Gabon [Laurance *et al.*, in press, 2004] consumed 33 and 80 tons of wildlife meat per year, respectively. For exploited wildlife, the ecological impacts of logging and hunting are much greater than that of logging alone.

Finally, even when protected from colonists and hunters, logging increases the vulnerability of forests to fire by rupturing the forest canopy and creating piles of dry, flammable debris [Holdsworth and Uhl, 1997]. In the Amazon and Borneo, millions of hectares of logged forest were destroyed by wildfires during the 1982–83 and 1997–98 El Niño droughts [Nepstad *et al.*, 1999; Siegert *et al.*, 2001; Cochrane, 2003]. In this way, logging can seriously increase forest destruction.

##### *Habitat Fragmentation and Its Synergisms*

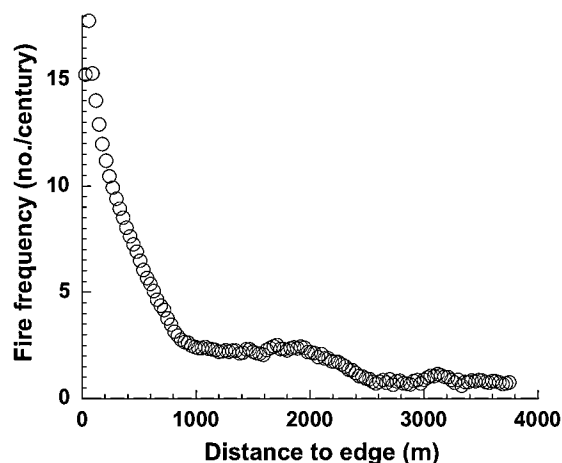
In most human-dominated landscapes, fragmented forests are not merely reduced and isolated; they are also affected by other perturbations that may interact additively or synergistically with fragmentation [Laurance and Cochrane, 2001]. In agricultural and urban areas, acid rain, pesticides and herbicides, hydrological changes, livestock grazing, and pressure from invading species can degrade the ecology of forest fragments [Myers, 1987; Hobbs and Hueneke, 1992; Weathers *et al.*, 2001]. Tropical forest fragments and isolated nature reserves are frequently affected by selective logging and fuelwood gathering [Curran *et al.*, 1999, 2004; Oluput and Chapman, in press, 2004; Peres and Michalski, in press, 2004]. One of the most widespread threats to wildlife in fragmented tropical forests is hunting. Many local and regional extinctions of vertebrates in fragmented landscapes of the Brazilian Atlantic forest [Chairello, 1999; Cullen *et al.*, 2000], Amazonia [Peres and Michalski, in press, 2004], Mesoamerica [Glanz, 1991; Wright *et al.*, 2000], Madagascar [Smith, 1997], West Africa [Brashares *et al.*, 2001], and Southeast Asia [Corlett and Turner, 1997; Pattanavibool and Dearden, 2002] have been attributed to over-hunting.

An especially destructive synergism in fragmented tropical landscapes is the interaction between habitat fragmentation and surface fires [Kauffman and Uhl, 1991; Cochrane *et al.*, 1999; Gascon *et al.*, 2000]. This occurs because forest remnants have dry, fire-prone edges, are juxtaposed with frequently burned pastures or slash-and-burn farming plots, and

are often degraded by selective logging, which increases forest desiccation and fuel loading. In a fragmented landscape in eastern Amazonia, surface-fire incidence was significantly increased (Figure 4) [AC4] within 2400 m of forest edges and sharply elevated within 600–800 m of edges [Cochrane and Laurance, 2002]. Because rainforest vegetation is especially vulnerable to burning, a single surface fire can cause an extensive dieback of forest margins whereas recurring fires may lead to fragment “implosion” [Gascon *et al.*, 2000]. Given the irregular shapes of most forest fragments, even large (>100,000 ha) fragments could be vulnerable to edge-related fires, particularly in more seasonal areas of the tropics [Cochrane and Laurance, 2002]. Given present patterns of forest fragmentation, 45 million ha of forest may currently be vulnerable to edge-related fires in the Brazilian Amazon alone [Cochrane, 2001]. Thus, habitat fragmentation sharply increases the vulnerability of forests to fire, with destructive effects on tropical ecosystems and wildlife.

#### *Additional Synergisms*

I have briefly described some of the most important ecological impacts of land-use change on tropical fauna and plant communities. It is beyond the scope of this chapter to address the impacts of large-scale atmospheric and climatic changes on tropical biota [cf. Malhi and Phillips, 2004]. It must be emphasized, however, that such phenomena are likely to interact with local land-use changes, posing potentially serious threats to biota in some regions. These are some of the most poorly understood of all environmental synergisms. Will global warming increase the frequency of El Niño-related droughts [Timmermann *et al.*, 1999], leading to more frequent tropical



**Figure 4.** Estimated fire frequencies as a function of distance from forest edge for a fragmented landscape (Tailândia) in eastern Amazonia. Data are based on remote-sensing imagery spanning a 14-year period [after Cochrane and Laurance, 2002].

wildfires? Will climatic changes threaten cool-adapted montane species [Pounds *et al.*, 1999; Williams *et al.*, 2003] and those whose geographic ranges have been disrupted by habitat loss and fragmentation [Thomas *et al.*, 2004]? Will large-scale deforestation lead to declines in regional evapotranspiration and rainfall in tropical regions [Lean and Warrilow, 1989; Lawton *et al.*, 2001]? At this stage only one point is certain: the world is changing very rapidly, with potentially critical implications for tropical ecosystems and biota.

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