Positive Feedbacks among Forest Fragmentation, Drought, and Climate Change in the Amazon

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Abstract: *The Amazon basin is experiencing rapid forest loss and fragmentation. Fragmented forests are more prone than intact forests to periodic damage from El Niño–Southern Oscillation (ENSO) droughts, which cause elevated tree mortality, increased litterfall, shifts in plant phenology, and other ecological changes, especially near forest edges. Moreover, positive feedbacks among forest loss, fragmentation, fire, and regional climate change appear increasingly likely. Deforestation reduces plant evapotranspiration, which in turn constrains regional rainfall, increasing the vulnerability of forests to fire. Forest fragments are especially vulnerable because they have dry, fire-prone edges, are logged frequently, and often are adjoined by cattle pastures, which are burned regularly. The net result is that there may be a critical "deforestation threshold" above which Amazonian rainforests can no longer be sustained, particularly in relatively seasonal areas of the basin. Global warming could exacerbate this problem if it promotes drier climates or stronger ENSO droughts. Synergisms among many simultaneous environmental changes are posing unprecedented threats to Amazonian forests.*

Retroalimentaciones Positivas entre Fragmentación de Bosques, Sequía y Cambio Climático en el Amazonas

Resumen: *La cuenca del Amazonas esta sujeta a una rápida pérdida y fragmentación de bosques. Los bosques fragmentados son más propensos que los bosques intactos a daños periódicos por las sequías de El Niño, las que causan un incremento en la mortalidad de árboles, especialmente cerca de los bordes de bosque. Más aún, las retroalimentaciones positivas entre pérdida de bosque, fragmentación, fuego y cambio climático regional son muy probables. La deforestación reduce la evapotranspiración de plantas, que a su vez constriñe la precipitación pluvial regional, incrementando la vulnerabilidad de los bosques al fuego. Los fragmentos de bosque son especialmente vulnerables porque tienen bordes secos, propensos al fuego, son talados frecuentemente y a menudo son contiguos a pastizales que se queman regularmente. El resultado neto es que puede haber un "umbral de deforestación" crítico sobre el que las selvas de la Amazonía ya no pueden sostenerse, particularmente en áreas relativamente estacionales de la cuenca. El calentamiento global podría exacerbar este problema si promueve climas más secos o sequías de El Niño más fuertes. Los sinergismos entre muchos cambios ambientales simultáneos son amenazas sin precedentes para los bosques de la Amazonía.*

Introduction

The Amazon basin sustains almost 60% of the world's remaining tropical rainforest and plays crucial roles in biodiversity conservation, carbon storage, and regional hydrology and climate (Salati & Vose 1984; Fearnside 1999). It is also experiencing the world's highest absolute rate of forest destruction, averaging roughly 3–4 million ha per year (cf. Whitmore 1997; Laurance et al. 2001).

The rapid pace of deforestation has several interrelated causes. Human populations in the Amazon have increased sharply in recent decades as a result of immigration from other areas and high rates of intrinsic growth. Industrial logging and mining are growing dramatically

in importance, and expanding road networks are increasing access to forests for slash-and-burn farmers, ranchers, and hunters. The spatial patterns of forest loss are also changing: past deforestation has been concentrated in the eastern and southern areas of the Amazon, but new highways, feeder roads, and colonization and logging projects are now penetrating into the heart of the basin (Laurance 1998; Carvalho et al. 2001; Laurance et al. 2001). Finally, human-caused wildfires are becoming an increasingly important cause of forest destruction (Fearnside 1995; Barbosa & Fearnside 1999; Cochrane et al. 1999; Nepstad et al. 1999*a*).

Climatic variability is also having important effects on Amazonian forests. Large expanses of the basin have strong dry seasons that are exacerbated by El Niño–Southern Oscillation (ENSO) droughts, which occur at 3- to 7-year intervals and greatly increase the vulnerability of forests to fire (Nepstad et al. 1999*a*). We describe some effects of a strong ENSO drought on fragmented forests in the central Amazon, then highlight the potential for positive feedbacks among forest loss, fragmentation, fire, and regional climate change. We argue that the synergistic effects of forest conversion and climatic variability pose unprecedented threats to Amazonian forests.

Droughts and Fires in the Amazon

Under natural conditions, fire is a rare event in Amazonian rainforests (Sanford et al. 1985; Saldariagga & West 1986). Over the past two millennia, major incursions of fire into intact forests appear to be associated mainly with exceptionally severe ENSO droughts, occurring at roughly 400- to 700-year intervals (Meggers 1994). Consequently, most plants and animals are poorly adapted to fire. The large majority of tree species, for example, possess thin bark and can be killed by even low-intensity surface fires (Uhl & Kauffman 1990; Kauffman 1991; Cochrane & Schulze 1999).

The climate of the Amazon basin is far from uniform. In general, the southern, eastern, east-central, and north-central areas are driest, with lower annual rainfall and stronger dry seasons than the more westerly parts of the basin. Evergreen rainforests in seasonally dry areas persist only by having deep root systems (>8 m) that access groundwater during the dry season. About half of the closed-canopy forests in the Brazilian Amazon require deep roots for survival (Nepstad et al. 1994, 1996).

The drier areas of the Amazon are most likely to suffer depleted soil-water reserves during ENSO droughts, causing hydric stress and increased leaf shedding in plants. Although rainforests are normally almost impenetrable to fire, leaf litter accumulates during droughts and becomes drier because of increased canopy openness and understory insolation. This increases the likelihood of

ground fires—especially if the forest has been logged, creating additional canopy openings and woody debris. Studies integrating data on seasonal soil-water availability, recent fires, and logging activity show that roughly 200,000 km^2 of closed-canopy forests in the Brazilian Amazon become vulnerable to fire during normal years (Nepstad et al. 1998, 1999*a*). This figure could approach 1.5 million km^2 during major droughts (D. C. Nepstad, oral presentation, first annual scientific conference of the National Aeronautics and Space Administration Long-Term Biosphere-Atmosphere Experiment in the Amazon [NASA-LBA]).

Droughts and Fragmentation

Pace of Fragmentation

Forest fragmentation is affecting increasingly large expanses of the Amazon. By 1988, the area of the Brazilian Amazon that was fragmented into blocks of <100 km^2 or was prone to edge effects (<1 km from the nearest clearing) was over 150% larger than the area actually deforested (Skole & Tucker 1993). Given that over 14% of the Brazilian Amazon has now been deforested, the total area affected by fragmentation, deforestation, and edge effects may comprise one-third of the region today (Laurance 1998). This figure would rise further if the roughly 10,000–15,000 km^2 of forest affected each year by legal and illegal logging were included (Nepstad et al. 1999*b*). Many forest fragments are logged, altering forest structure and microclimate and further increasing the forest's vulnerability to fire (Uhl & Buschbacher 1985; Laurance et al. 2000*b*).

Drought-Induced Tree Mortality

The 1997 ENSO resulted in an unusually strong dry season (June–October) throughout much of the Amazon basin. In the central Amazon, rainfall recorded in the 1997 dry season (232 mm) was less than one-third of normal (745 ± 128 mm), and the number of days without rain nearly doubled, from an average of 57 to 102 (Laurance et al., in press). The effects of the 1997 ENSO drought were assessed on fragmented and continuous forests in central Amazonia, near Manaus, Brazil, based on long-term data on tree mortality collected before, during, and after the drought (Williamson et al. 2000; Laurance et al., in press). For 23 permanent, 1-ha plots, we compared annualized mortality rates of trees at least 10 cm diameter at breast height (dbh) between a 5- to 17-year interval before the drought and a 12- to 16-month period during the drought (Fig. 1). Twelve of the plots were located in forest interiors (230–1700 m from the nearest edge), with 11 plots established near forest
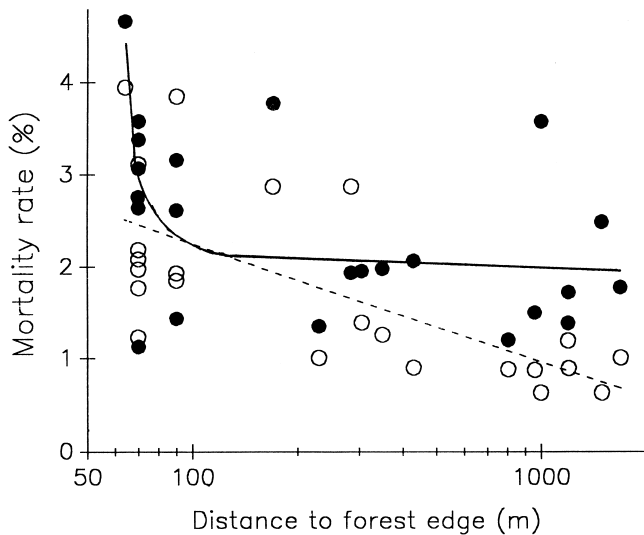
Figure 1. Relationship between distance of plots to the nearest forest edge and mortality rates of Amazonian trees for both predrought (open circles) and El Niño–Southern Oscillation (ENSO) (filled circles) intervals. Lines show exponential curves fitted to the data (baseline, dotted line; ENSO, solid line). Linear regressions comparing observed and fitted values were highly significant in both cases (predrought, $F_{1,21} = 18.87$, $R^2 = 47.3\%$, $p = 0.0003$; ENSO, $F_{1,21} = 10.78$, $R^2 = 33.9\%$, $p = 0.004$).

edges (plot center 60–170 m from edge). Even in normal years, microclimatic changes and increased wind turbulence cause elevated tree mortality and a loss of living forest biomass within 100–300 m of fragment edges (Laurance et al. 1997, 1998*a*, 1998*b*, 2000*a*).

During the drought, mortality rates increased in 20 of the 23 plots. Tree mortality rose both in forest interiors (11 of 12 plots) and near edges (9 of 11 plots), and in both cases these differences were significant ($p = 0.003$ and $p = 0.03$, respectively; sign tests). Mean annual mortality increased from 2.44% to 2.93% near edges and from 1.13% to 1.91% in interiors. When we plotted the baseline and drought mortality rates as a function of distance to forest edge, using a three-parameter exponential model, we found that mortality during the drought increased throughout the forest but most dramatically within 50–70 m of edges (Fig. 1).

Characteristics of Dying Trees

Few differences were detected among trees that died before and during the drought, suggesting that most central Amazonian trees are similarly vulnerable to drought (Table 1). When edge and interior plots were analyzed separately, no significant differences ($p > 0.15$) were found in the proportions of major tree families, successional guilds, or tree-size categories among trees that died be-

fore and during the drought. There was a significant difference, however, in the proportions of species that died most frequently in interior plots ($p = 0.016$), principally because individuals of the large palm *Oenocarpus bacaba* died more often than expected during the drought. There was no difference on edge plots (Table 1).

When the predrought and drought intervals were pooled, the proportions of dying trees in different successional guilds differed between forest edges and interiors ($p = 0.052$), principally because a higher fraction of pioneer trees died on edges than interiors (Table 1). In

Table 1.   Characteristics of Amazonian trees in forest edges and interiors that died before and during an El Niño–Southern Oscillation drought (chi-square tests).

Category	χ^2	df	p
Predrought vs. drought intervals			
proportions of common tree families (>40 dead trees overall)			
edge plots (13 families)[a]	12.97	12	0.37
interior plots (9 families)[b]	8.13	8	0.42
proportions of four successional guilds[c]			
edge plots	4.76	3	0.19
interior plots	3.00	3	0.39
proportions of two successional guilds[d]			
edge plots	0.87	1	0.35
interior plots	0.00	1	0.96
frequently dying species (>15 dead trees overall)			
edge plots[e]	13.24	15	0.58
interior plots[f]	8.22	2	0.016
tree-size categories[g]			
edge plots	5.43	4	0.25
interior plots	1.36	4	0.85
overall comparisons between forest edges and interiors			
four successional guilds[c]	7.74	3	0.052
two successional guilds[d]	0.00	1	0.96
tree-size categories[g]	12.15	4	0.016

[a]*Annonaceae, Arecaceae, Burseraceae, Chrysobalanaceae, Euphorbiaceae, Lauraceae, Lecythidaceae, Leguminosae, Melastomataceae, Moraceae, Myristicaceae, Sapotaceae, Violaceae.*
[b]*Annonaceae, Arecaceae, Burseraceae, Chrysobalanaceae, Lecythidaceae, Leguminosae, Lauraceae, Moraceae, Sapotaceae.*
[c]*Pioneer families (Boragaceae, Cecropiaceae, Melastomataceae, Malpighiaceae, Sterculiaceae), early successional families (Annonaceae, Clusiaceae, Euphorbiaceae, Proteaceae, Rubiaceae), late-successional families (Burseraceae, Elaeocarpaceae, Lauraceae, Meliaceae, Moniaceae, Sapindaceae), old-growth families (Chrysobalanaceae, Lecythidaceae, Sapotaceae, Voclysaceae).*
[d]*Pioneer/early successional guild versus late-successional/old-growth guild (see footnote c for families within guilds).*
[e]*Sixteen species died frequently enough on edges to permit analysis: Eschweilera spp. 2, Euterpe precatoria, Mabea caudata, Miconia burchellii, Ocotea amazonica, Oenocarpus bacaba, Paramachaerium ormosioides, Protium apiculatum, P. decandrum, P. grandifolium, P. cf. llewelynii, Rinorea flavescens, Scleronema micranthum, Tachigali plumbea, Virola calophylla, V. sebifera.*
[f]*Three species died frequently enough in interiors to permit analysis: Oenocarpus bacaba, Protium hebetatum, Eschweilera coriacea.*
[g]*Diameter classes: 10–15, 15.1–20, 20.1–30, 30.1–59.9, and 60 cm.*

addition, many more large trees (at least 60 cm in diameter) died on edges than expected ($p = 0.016$).

Although there is little evidence that the ENSO drought caused strongly biased patterns of tree mortality, our comparisons are conservative because of limited sample sizes. Over 85% of the tree species in the study area are rare (mean density <1 stem/ha), and our analyses were limited to trees that died, reducing our ability to detect differences in mortality among taxa and ecological groups. Moreover, the forest edges in our study area were at least 14 years old at the time of the drought and had been "sealed" to some extent by dense secondary vegetation that probably reduced the penetration of hot, dry conditions into the forest (cf. Kapos et al. 1997; Didham & Lawton 1999; Mesquita et al. 1999). Fragmentation-induced changes in physiological tolerance and floristic composition, such as a possible increase in drought-tolerant, semideciduous species (Condit et al. 1995, 1996) may also have "preadapted" our forest edges to drought conditions. Drought effects on recently fragmented forests (<5 years) are likely to be greater than those observed here because their edges are more open and the vegetation is poorly acclimated to edge conditions.

Other Drought Effects

In September and October of 1997, W.F.L. surveyed about 25 transects (100–200 m in length) perpendicular to the edges of a 10-ha forest fragment and a 40-m-wide road that bisected a large forest tract (with varying edge aspects); he observed that leaf shedding by drought-stressed trees had increased sharply, especially within 50–60 m of forest edges. Leaf litter near edges became exceptionally dry and brittle as foliage density in the canopy was reduced, allowing more sunlight into the understory. A general accumulation of tree limbs and fine woody debris occurred near edges, probably as a direct result of increased tree mortality. Thus, forest edges became exceptionally flammable during the drought, with a large concentration of dry fuels. When an ignition source is present, such conditions can lead to groundfire (Kauffman & Uhl 1991; Cochrane et al. 1999; Nepstad et al. 1999b) that can penetrate as far as several kilometers into forests (Cochrane 2001 [this issue]). Although of low intensity, these fires kill many small trees, creating canopy openings and a subsequent rain of woody debris that greatly increase the chances of catastrophic wildfires in the future (Cochrane et al. 1999).

Fortunately, there were few ignition sources near our fragments because our study area is well protected, but illegal hunters started several fires along nearby roads. These fires mainly burned regrowth trees along road edges, although they also occasionally penetrated into the adjoining primary forest. In most fragmented Amazonian landscapes, the incidence of accidental and purpose-ful fires is far greater than in our experimental study area (e.g., Nepstad et al. 1999a; Cochrane 2001 [this issue]).

The increased mortality and subsequent decomposition of trees in forest fragments could be a significant source of atmospheric carbon emissions (Laurance et al. 1997, 1998b). In addition to accelerating tree mortality, the drought may have generated further carbon emissions because forest productivity obviously fell because of reduced soil moisture (cf. Raich et al. 1991) and because plant respiration rates may have increased as a result of elevated temperatures (Grace et al. 1995). For these same reasons, even intact Amazonian forests apparently change from carbon sinks to carbon sources during ENSO droughts (Tian et al. 1998), but such effects may well be stronger in fragmented forests, which are more exposed to hot, desiccating conditions.

Positive Feedbacks with Climate Change

The processes of forest conversion, fire, and climatic change in the Amazon appear to reinforce one another in an alarming process of positive feedback (Fig. 2). Deforestation leads to forest fragmentation and the creation of fire-prone habitats such as cattle pastures, which are burned periodically to control weeds, and regrowth forests, which are far more flammable than primary forest. These modified habitats can carry fire after only a few days of dry weather (Uhl & Kauffman 1990). As described above, fragments of primary forest are also susceptible to fire because their edges contain abundant fuel and are likely to become desiccated during prolonged dry weather (Gascon et al. 2000; Cochrane 2001 [this issue]). Logging also increases the vulnerability of forests to fire and interacts synergistically with fragmentation because logged fragments are exceptionally prone to edge-related fires.

On a regional scale, deforestation reduces rainfall in two ways (Fig. 2). First, water vapor produced by forests through evapotranspiration contributes substantially to Amazonian rainfall (Salati & Vose 1984). Large-scale deforestation could cause an estimated 20% decline in Amazonian rainfall, leading to lower humidity, higher surface temperatures, and more severe dry seasons (Lean & Warrilow 1989; Shukla et al. 1990). Second, smoke from forest fires can reduce rainfall and possibly cloud cover by trapping moisture and inhibiting the formation of raindrops (Rosenfeld 1999; Ackerman et al. 2000). Thus, by promoting regional climate change, deforestation leads to greater drought stress and still more fires and forest conversion.

This positive feedback process becomes especially important during ENSO droughts, when even intact forests can become seriously stressed by drought (Nepstad et al. 1999a; Williamson et al. 2000). Modest changes in
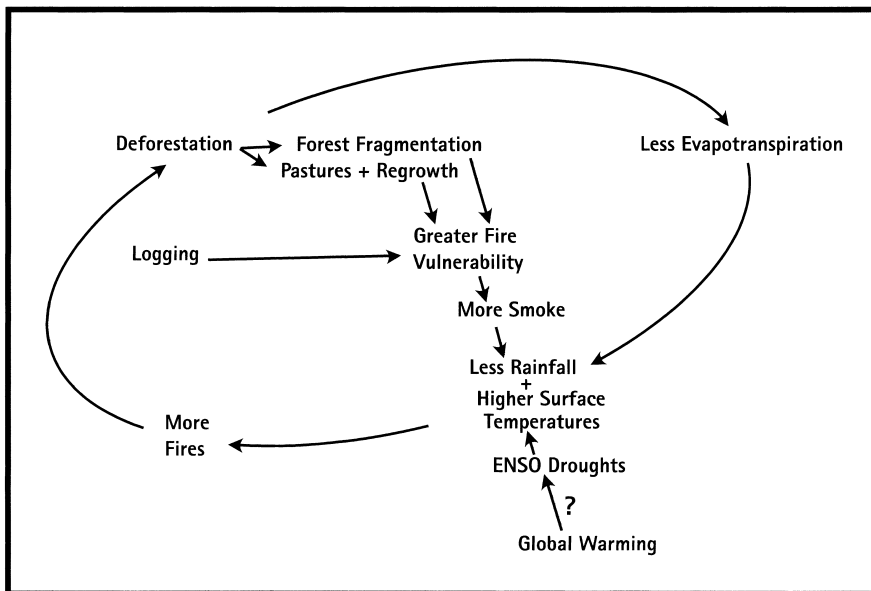
Figure 2. Positive feedbacks among forest fragmentation, logging, fires, and climate change in the Amazon (ENSO, El Niño–Southern Oscillation).

rainfall during the critical dry-season months can sharply increase the likelihood of ground fires and larger, more destructive wildfires. In the Brazilian state of Roraima, for instance, drought-induced fires destroyed over 1.1 million ha of fragmented and intact forest in 1998 (Barbosa & Fearnside 1999). The potential for wildfire is further increased by the prevalent use of fire in the Amazon. During a 4-month period in 1997, for example, a weather satellite detected almost 45,000 separate fires in the Amazon, most of which were ignited by ranchers and slash-and-burn farmers (Brown 1998).

In the future, catastrophic wildfires are most likely to increase in the southern, eastern, central, and north-central areas of the Amazon, where human population pressures are greatest and rainforests are most seasonal. When deep groundwater is depleted during droughts, these forests become far more prone to burning, especially when logged or fragmented (Nepstad et al. 1999*a*, 1999*b*). Because of positive feedbacks among deforestation, fragmentation, regional drying, and wildfires, deforestation above some critical threshold could make it difficult or impossible to sustain rainforests in seasonal areas (Fig. 2). The nature of this threshold is difficult to predict but will probably depend on factors such as local climatic conditions, soils, and prevailing land-use practices. It is important to emphasize that even in areas with limited deforestation (<20%), much of the remaining forest is altered by fragmentation, logging, ground fires, hunting, wildcat mining, edge effects, and other ecological changes (Lovejoy et al. 1986; Skole & Tucker 1993; Dale et al. 1994; Laurance et al. 1997, 1998*a*, 2000*a*; Cochrane et al. 1999; Cochrane 2001 [this issue]; Nepstad et al. 1999*b*; Laurance 2000).

In addition, global climate change may exacerbate current threats to Amazonian forests (Fig. 2). An increasing

body of evidence suggests that extreme weather events, such as ENSO droughts and tropical storms, may increase in frequency or severity as a result of global warming (Intergovernmental Panel on Climate Change 1996; Timmerman et al. 1999). At the least, the frequency of warm-weather events should rise and the likelihood of cool-weather events decline as a consequence of higher mean temperatures (Mahlman 1997). The net effect, if deforestation, fragmentation, and logging continue apace, is that rainforests could become unsustainable across large expanses of the Amazon basin.

Implications

As large-scale clearing and forest fragmentation proceed, it will become increasingly difficult to maintain Amazonian reserves and semiprotected areas such as national forests and indigenous lands because of the contagious and uncontrolled spread of fires, logging, and regional climate change. Currently, many protected areas in the Amazon are little more than "paper parks" with inadequate protection (but see Bruner et al. 2001). A recent analysis of 86 federal parks and protected areas in Brazil, for example, found that 43% were at high to extreme risk as a result of illegal deforestation, colonization, hunting, isolation of the reserve from other forest areas, and additional forms of encroachment. Of all reserves, 55% were judged to have nearly nonexistent management (Ferreira et al. 1999). In Brazil's Pará state, nearly three-quarters of all protected areas are already physically accessible to loggers (Verissimo et al. 1998).

As human populations continue to expand in the Amazon, the already serious problems of illegal logging and

forest clearing are likely to worsen. Illegal logging is a critical threat to forests because of the direct effects of loggers and the secondary effects of hunters and slash-and-burn farmers, who use networks of logging roads to gain access to remote frontier areas (Uhl & Buschbacher 1985; Nepstad et al. 1999*a*). The Brazilian government recently estimated that 80% of all timber cutting in the Amazon is illegal (Laurance 1998). Cattle ranching and government-sponsored colonization projects are also major causes of forest destruction (Fearnside 1993; Nepstad et al. 1999*a*).

Without fundamental changes in prevailing land-use practices and development policies, wildfires almost certainly will become more common in the Amazon. Primary forests play a key role as firebreaks (Nepstad et al. 1996), and as deforestation and fragmentation increase, the prospects for fires to rage unimpeded across large areas could rise sharply. Over the last two millennia, large Amazonian fires have occurred only during rare mega–ENSO droughts (Meggers 1994). Today, however, a growing concern is that, as a direct consequence of rapid forest degradation, lesser but far more frequent ENSO events (such as the 1983 and 1997 droughts) could have equally serious consequences. Forest burning is a major source of greenhouse gases (Houghton 1991; Fearnside 2000), and increasing carbon emissions from Amazon forest destruction would contribute significantly to global warming.

Unfortunately, the already high rate of Amazonian forest conversion is likely to accelerate. Under the auspices of its Avança Brasil (Advance Brazil) program, the Brazilian government intends to invest about $40 billion over the next several years in Amazonian infrastructure, including many new hydroelectric dams, power lines, gas lines, railroads, and river-channelization projects (Carvalho et al. 2001; Laurance et al. 2001). The current network of paved highways will be roughly doubled, by some 7500 km, providing year-round access to vast new frontiers for loggers, ranchers, miners, and colonists. If this program proceeds as planned, substantial increases in the rate and extent of forest conversion and fragmentation would be virtually unavoidable (Laurance et al. 2001). As a consequence, negative synergisms among Amazonian forest fragmentation, fire, and regional climate change could pose an even more serious threat in the future.

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