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FIRE IN AMAZONIAN SELECTIVELY LOGGED RAIN FOREST AND THE POTENTIAL FOR FIRE REDUCTION

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Abstract. Approximately 8000 km² of Brazilian Amazon forest are selectively logged each year. Although virgin forest in eastern Amazonia is generally immune to fire, selectively logged forests are susceptible to fire. In eastern Amazonia we employed permanent-plot studies, forest fuel moisture measurements, and hemispheric canopy photographs to study the impacts of fire on a selectively logged forest, the microclimatic conditions that foster forest fires, and the measures that loggers might take to reduce fire incidence.

Significant tree mortality followed a typical ground fire in a selectively logged forest. Forty-four percent of all trees $\geq$10 cm in diameter at breast height died in a burned plot while only 3% died in an unburned plot. In large logging gaps the density of regenerating pioneer species increased by >60% in burned plots 15 mo after the fire, while it decreased by >40% in unburned plots.

The rate of fuel drying in selectively logged forest was influenced by photon flux density (PFD), time since logging, and logging techniques. There was a significant (P = 0.005) negative correlation between PFD and the number of days fuel sticks required to reach the point where fire could spread. In a recently logged forest, large logging gaps (>700 m²) reached fire susceptibility after 6 d, and medium-sized logging gaps ($\approx$200–700 m²) reached fire susceptibility after 15 d. But fire susceptibility declines over time as logging gaps become densely packed with saplings; fuel moisture conditions in large gaps of forest selectively logged 4 yr earlier were similar to those found in virgin forest, thus reducing the likelihood of fire. Careful logging also can reduce the likelihood of fire. Special low-impact logging techniques remove the same amount of timber as do the more typical high-impact logging techniques, but fire is significantly less likely because the more careful operation avoids the creation of large logging gaps, the most fire susceptible areas.

Key words: Amazonian rain forest and selective logging; fire damage and tree mortality; fire reduction in selectively logged forests; fire susceptibility, effects of logging methods on; forest fragmentation, fire-induced; gap size and fuel moisture in logged forests; hemispheric canopy photography; photon flux density and forest fuel moisture; tree and vine/liana regeneration after fire.

INTRODUCTION

For thousands of years fire has influenced the structure and composition of Amazonian forests. In northwestern Amazonia soil charcoal indicates the occurrence of multiple fires in the last 6000 yr (Sanford et al. 1985) that appear to have affected forest structure in the region (Saldarriaga and West 1986). Nelson (1994) suggested that large natural burns may explain the unusual structure of eastern Amazonian liana forests. In past epochs, prolonged droughts induced by mega-Niño events (occurring on the order of hundreds of years) together with more frequent shorter droughts associated with El Niño events (occurring on the order of tens of years) have created conditions that probably would have permitted extensive fires in Amazonia (Meggars 1994).

In the last 30 yr some areas of Amazonia have undergone rapid land-use changes (Hecht 1981). In Pará State of eastern Amazonia, roads and colonization activities (ranching, farming, logging) have led to the fragmentation of the seasonally dry evergreen forest. Like climatic changes in the distant past, these land-use changes have contributed to an increased incidence of fire in the region. During the dry season, when average rainfall is <50 mm/mo and rainless periods >16 d occur 1–7 times every year (Empresa Brasileira de Pesquisa Agropecuária and Woods Hole Research Center, *unpublished data*), pasture grasses often become water stressed, leaf area declines (Nepstad et al. 1994), and pasture fire becomes possible after only a single rainless day (Uhl and Kauffman 1990). Since fire is the principal tool for land clearing and pasture maintenance (Fearnside 1990), ignition events are common. The satellite studies of Setzer and Pereira (1991) reported the

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burning of 2.0×10^5 km² (≈5%) of the Legal Brazilian Amazon in 1987 alone.

In recent years selective logging has become the most important land-use activity in the region and is most likely to increase in importance (Verissimo et al. 1992). It is estimated that ≈8000 km² (≈0.2%) of the Brazilian Amazon are selectively logged annually (IMAZON [Instituto do Homen e Meio Ambiente da Amazonia], *unpublished data*). Although loggers often only extract 4–8 trees/ha, the resulting forest is fragmented into a mosaic of gaps and forest patches and canopy cover is reduced by about half (Uhl and Vieira 1989). Selective logging transforms fire-immune primary forest into fire-susceptible forest. Post-logging fuel loads in logged forests are three times higher than in uncut primary forest, and large gaps can burn after only 5–6 rainless days in the dry season (Uhl and Kauffman 1990). Uhl and Buschbacher (1985) reported that fire had occurred in the logged forests of 50% of the ranchers interviewed ($n = 15$). This phenomenon seems to be continuing. In 1994, 40% of the ranchers indicated that areas of their logged forests burned between 1987 and 1992 ($n = 14$) (A. Holdsworth, *unpublished data*). The resulting fires can be large; in 1988 a single fire burned ≈1000 km² of logged forest (D. Nepstad, *personal communication*). Finally, the fires are not unique to the Amazon; for example, in 1982–1983 1.2×10^6 ha of selectively logged forest burned in the Indonesian province of East Kalimantan (Leighton and Wirawan 1986).

Most fires are intentionally set in pastures and fields from which they then inadvertently spread into nearby selectively logged forests. In forest patches of the logged forest the ground fires spread relatively slowly, with mean flame heights of only 10–30 cm. When the fire reaches logging gaps and the high fuel loads of logging slash, flame heights typically increase to several meters, yet rarely (if ever) reach into the crowns of trees (A. Holdsworth, *personal observation*).

Fires in pastures burn through land that is already cleared, thus having little effect on biodiversity. But fire in logged forest could have much greater effects on biodiversity as well as the structure and composition of regenerating forest. Following drought and fire in logged forests of Malaysia, Woods (1989) recorded 38–94% (depending on size class) mortality of trees ≥10 cm diameter at breast height (dbh) and an increase in the understory density of woody creepers, invasive grasses, and ferns. Two percent and 96%, respectively, of all stems ≥10 cm dbh were killed after ground fires spread through closed canopy and open canopy selectively logged forests in the eastern Amazon (Uhl and Buschbacher 1985). Kauffman (1991), working in the same region, found that tree mortality depended on fire severity, and ranged from 36 to 69%.

While the studies cited above have documented some effects of fire on forest structure and composition, few have suggested how these effects could be reduced. We hypothesize that careful logging practices that reduce logging-inflicted damage to unharvested trees and foster the natural regeneration of the forest could reduce the risk of fire to selectively logged forests. An experimental logging study in the eastern Amazon demonstrated that through pre-logging vine-cutting, skid-trail planning, and directional felling the amount of canopy lost was low compared to that in high-impact logging operations (Johns et al., *in press*). Hence, if fire susceptibility is correlated with canopy loss, the careful execution of timber extraction could reduce the probability of forest fires in eastern Amazonia.

In this study we address the following three questions:

1) What were the impacts of a fire on the structure and composition of a seasonally dry selectively logged forest in the eastern Amazon?

2) How do fuel dry-down rates differ among micro-environments of a selectively logged forest and how are these rates related to incoming solar radiation?

3) Do improved logging techniques reduce canopy loss and thus make the forest more resistant to invasion by fire?

Methods

Study region and sites

This study occurred at two ranches (each >5 000 ha), Fazenda Vitoria and Fazenda Sete, located 35 km apart near Paragominas, Pará State, Brazil (3° S, 48° W) (Fig. 1). Aided by government incentives, conversion of forest to pasture in Paragominas County began in the early 1970s, but diminishing forage productivity led to the abandonment of some of these pastures after 5–10 yr. Abandonment was followed by the development of second-growth forest (Nepstad et al. 1991). Recently, selective logging has become the dominant activity in Paragominas County; in 1970 only one sawmill existed in the 2.49×10^6 ha county, while in 1992 137 mills were operating (Verissimo et al. 1992). Most of the selective logging that supplies these mills occurs in forests owned by ranchers.

At both Fazenda Vitoria and Fazenda Sete the soils are predominantly oxisols and the mature forest has a canopy height of 25–35 m (Uhl et al. 1988). Both properties have a combination of pasture, second-growth, selectively logged, and primary forests. Average annual rainfall is ≈1750 mm, with a notable dry season that occurs between July and December when average rainfall is <50 mm/mo (1973–1982 Empresa Brasileira de Pesquisa Agropecuária [Belem, Pará, Brazil] records). Topographic relief is slight at both sites.

Mechanized selective logging began in the early 1980s at Fazenda Vitoria. Our study area at Fazenda Vitoria was logged in 1989. Bulldozers and trucks created a network of roads and skid trails to remove an average of 3 trees (18 m³)/ha (Table 1). The creation of the roads and skid trails in conjunction with the cutting and extraction of the desirable trees directly

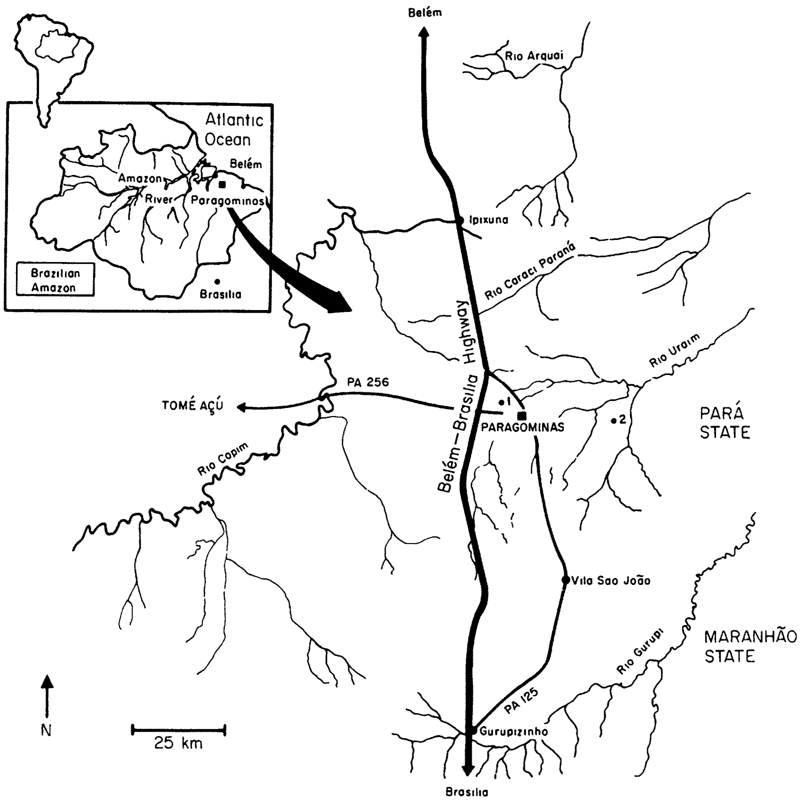

FIG. 1. Location of the two study areas near Paragominas in the eastern Amazon, Pará State, Brazil. 1 = Fazenda Vitoria; 2 = Fazenda Sete.

resulted in damage (uprooting, stem-breakage, bark removal) to ≈120 trees/ha and a 25% reduction in canopy cover (Verissimo et al. 1992).

Eight months after logging, a 50 × 1000 m (5-ha) plot was established adjacent to the main logging road to examine post-logging tree growth. The plot was divided into two 50 × 500 m (2.5 ha) sub-plots (Table 1). In one sub-plot (silviculture plot) vines >2 cm in diameter were cut and some non-commercial trees competing with future timber trees were girdled. The other sub-plot received no post-logging treatment. All trees ≥10 cm in diameter at breast height (dbh) were mapped, measured, and permanently marked in both sub-plots.

In November 1992 (3 yr after logging) a wildfire spread from a nearby pasture, through a second-growth forest, and into the two sub-plots. The fire occurred after 21 rainless days, part of an unusually strong dry season. The fire burned one sub-plot (no silviculture treatment) and ≈10% of the silviculture treatment plot before it stopped at a firebreak. It left many islands of unburned vegetation (Fig. 2B). The pre-existing plot data provided an excellent baseline for measuring changes in forest structure and composition following this fire, particularly given that there was no significant pre-fire difference between tree density by diameter class within the two sub-plots (unpaired t test; $t = 0.12$, $P = 0.91$).

To compare tree mortality with and without fire, the combined area of the original two sub-plots (5 ha) was divided into a 1.75-ha unburned control plot and a 2.75-ha burned plot. To eliminate any potential edge effects, all trees within 40 m of both sides of the fire boundary were excluded from the analysis of fire effects. This eliminated 0.25 ha each of burned and unburned area.

At the other site, Fazenda Sete, selective logging had occurred annually since the early 1980s. To study the benefits of a proposed low-impact logging system, a comparative demonstration study was conducted in 1993 (Table 1). In one 75-ha plot (725 × 1025 m), typical (high-impact) logging occurred. Low-impact logging occurred simultaneously in an adjacent 105-ha plot (750 × 1400 m). The high-impact logging removed 5.6 trees/ha (30 m³/ha) but created significantly larger canopy gaps than the low-impact logging, which extracted 4.5 trees/ha (37 m³/ha) (Johns et al., *in press*). The presence of forest tracts that had been selectively logged at different times in the past and the testing of low-impact logging techniques allowed us to examine how both time since logging and logging methods can affect fire susceptibility.

Fire effects on tree mortality and regeneration

To characterize the structure of the Fazenda Vitoria selectively logged forest following fire, all previously

TABLE 1. Dates of logging, fire, and plot measurements included at the two sites in this study.

Date	Event	Plot information
Fazenda Vitoria, 6.5 km northwest of Paragominas, Pará, Brazil		
Aug.–Sept. 1989	Logging of ≈150 ha	···
Oct. 1989	Gap regeneration plots (n = 34) established	1 × 3 m, 1 each in slash, no-slash, and skid-trail areas of 7 large gaps 1 × 3 m, 1 each in slash and no-slash areas of 7 small gaps
May 1990	Silvicultural treatment established	One 50 × 500 m (2.5 ha) silviculture sub-plot and one 50 × 500 m (2.5 ha) untreated sub-plot created
Oct. 1991	Gap regeneration plots re-censused	···
Nov. 1992	Fire burns 2.5 ha of untreated sub-plot and 0.5 ha of silviculture sub-plot	2.75 ha burned plot and 1.75 ha unburned plot created
Nov.–Dec. 1992	Trees in burned plot censused for burn damage	···
Jan. 1993, 1994	Gap regeneration plots re-censused	···
June 1994	Trees in burned and unburned plots re-censused	···
Fazenda Sete, 30 km east of Paragominas, Pará, Brazil		
Dec. 1993	Timber extraction using low-impact logging (105 ha) and high-impact logging techniques (75 ha) completed	···
Aug. 1994	Fuel moisture study in forests subjected to low-impact and high-impact logging techniques	···

marked trees ≥10 cm dbh were surveyed 1 mo and 30 mo after the fire. Trees were considered burned if the litter layer had burned to within 5 cm of their trunks. Mortality was defined as the death of all aboveground tissue.

To compare regeneration before and after fire, three 1 × 3 m plots were established in October 1989, 1 mo after logging in each of seven large gaps where ≥3 trees had been extracted. In addition, two 1 × 3 m plots were established in each of seven small gaps where 1 tree had been extracted (Table 1). In the seven large gaps, one plot was established where the bulldozer had maneuvered (referred to as "skid trail"), one in logging slash, i.e., fallen branches and debris ("slash"), and one in an area with no slash nor machine maneuvering ("no slash"). Small gaps only included slash and no-slash plots. All plants ≥10 cm in height were identified and permanently marked in October 1989 (the month of plot establishment) and re-measured in October 1991.

The fire (November 1992) burned approximately half of the gap plots. We re-surveyed all plots 15 mo after the fire, noting mortality, new establishment, and species (Table 1). For consistency, plots will be referred to as "burned" or "unburned" for measurements before and after the fire. Plants in the plots were grouped into five classes: non-timber trees and shrubs, vines and lianas, pioneer species, currently sawn timber species, and other species (unidentified individuals and forbs). The pioneer-species group consisted of the most common pioneer genera (*Cecropia, Banara, Vismia, Solanum,* and *Trema*). Currently sawn timber species include any of the nearly 90 species processed by sawmills in Paragominas County. A split-plot 3 × 2 factorial design ANOVA was used to test the interactions between the effects of ground treatment (slash, no slash, skid trail), burn (burned, unburned), and census time (1 yr before fire, 15 mo after fire, the time period of most concern) on density. Because only one fire burned all the plots, the burn treatment is pseudo-replicated, but the opportunity to evaluate change in species composition through time was worthwhile. Due to violations of the assumptions of ANOVA, density values were log transformed.

Fuel dry-down rates in a forest subjected to high-impact logging practices

To measure how fuel dry-down varies in different areas of a recently logged forest, 1-cm diameter, 10-h time-lag "fuel sticks" were suspended 25 cm above the ground in 20 locations in a 6-ha forest tract logged 8 mo earlier at Fazenda Sete. The sticks were placed in a range of light environments including closed-canopy forest patches, skid trails with a moderately open canopy, and large logging gaps (>700 m^2) with little canopy cover. Some stations (particularly those in large gaps) had a dense layer of knee-high regrowth overtopping the fuel sticks. When this occurred, we removed the leafy vegetation up to 1 m above the ground in a 1-m^2 area around the fuel sticks so that we would be measuring just tree canopy cover effects on fuel dry-down. To test the effects of 8 mo of regeneration on dry-down in three of the large gaps, we placed additional fuel sticks 50 cm into the ground-level regrowth. We also placed a fuel stick in 25 ha of uncut primary forest as a reference for the fuel sticks in the selectively logged forest. For 16 rainless days following a 1-cm rain at the beginning of the dry season in August 1994, we weighed all fuel sticks daily to ±0.1 g between 1300 and 1500, the period when air humidity levels are lowest (Uhl and Kauffman 1990). Moisture content was calculated on an oven-dry basis where per-

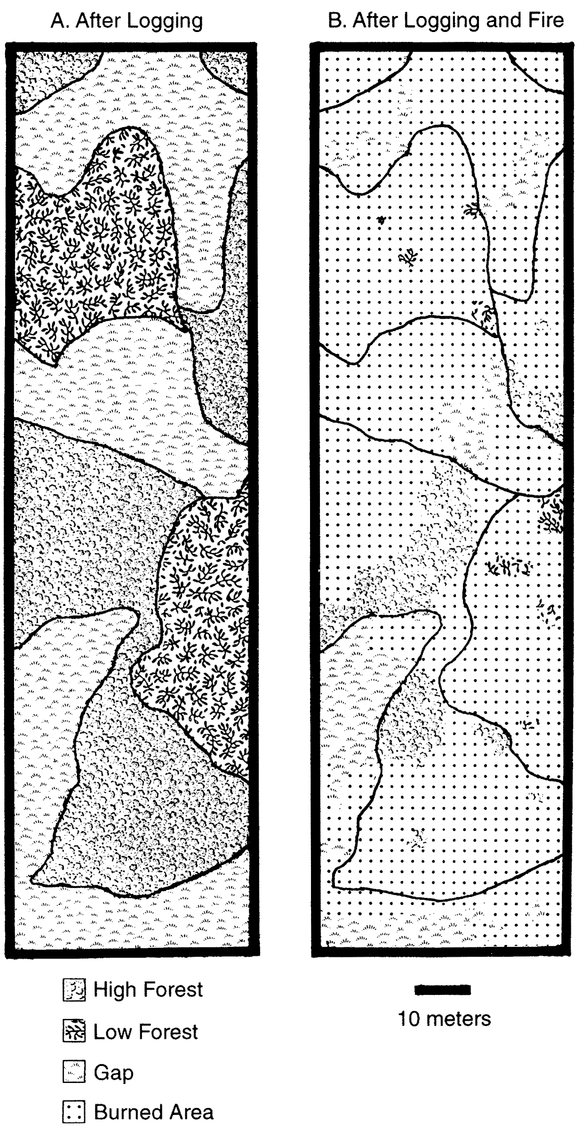

FIG. 2. Forest cover in a 50 × 200 m (0.1-ha) section of the Fazenda Vitoria burned plot near Paragominas, Pará, Brazil. Cover classes include high-forest (15–30 m high), low forest (6–15 m high and heavily vine laden), gap (areas where trees were extracted), and burned area (areas where fire burned the litter layer).

centage moisture content = [(field mass − oven-dry mass)/(oven-dry mass)] × 100.

To test for a relationship between fuel dry-down and the amount of direct solar radiation reaching each fuel stick, we took a hemispheric photograph of the canopy from 1 m above each fuel stick. Photographs were analyzed using the computer program SOLARCALC (Chazdon and Field 1987). Using the SOLARCALC calculations of the rate of direct photon flux density (PFD) arriving at each station and the number of days required for the fuel stick to cross the fire threshold (described below), we developed a regression model of fire susceptibility. For this model, a fuel-stick moisture content of 12% was assumed to be the threshold below which ignition of the litter layer and spread of fire is possible (Uhl and Kauffman 1990). Small fire trials during the measurements confirmed the validity of this threshold.

To measure the effect of 4 yr of regeneration on litter-layer dry-down in gaps where high-impact logging had occurred, we made paired fuel-stick moisture measurements in three large gaps four yr after logging. We removed the vegetation cover from a 10 × 10 m (100 m^2) area in the center of each gap to simulate the dry-down conditions found just after logging. One fuel stick was placed in the center of each cleared patch and another 10 m northwards, 5 m into the 3–5 m high regrowth. Fuel sticks were weighed daily during the 16 rainless-day period already mentioned.

Fire susceptibility of forests subjected to low-impact vs. high-impact logging practices

To measure the effects of logging techniques on fuel dry-down, we selected a 700 × 350 m (≈25 ha) plot in the forest subjected to high-impact logging and in the forest subjected to low-impact logging at Fazenda Sete. Both plots had similar pre-logging stand densities and roughly the same number of trees extracted. We randomly chose 10 of 16 possible 250-m transects along pre-existing lines 50 m apart. We then shot a hemispheric photograph every 50 m along all 10 transects in each 25-ha plot. Due to a technical malfunction, 12 photos (two adjacent transects) from the forest subjected to low-impact logging did not turn out, thus we re-shot them 7 mo later. To test for any seasonal changes in the canopy cover, we re-shot 11 stations in the high-impact logging area to compare with the photos of the same stations taken 7 mo earlier. There was not a significant difference between the photos from the two seasons (Mann-Whitney U test; $U = 50$, $P = 0.5$, $n = 11$ stations each season), thus we included the re-shot photos from the forest subjected to low-impact logging in the analysis. All photos were analyzed with SOLARCALC (Chazdon and Field 1987) to calculate the weighted percentage canopy openness (i.e., open sky points closer to vertical are weighted more than those closer to horizontal) and rate (number of minutes per day) of direct PFD (in units of $mol \cdot m^{-2} \cdot d^{-1}$) for each photo station. Applying the regression model of fire susceptibility developed from the fuel-stick measurements described above, we estimated the number of rainless days required to cross the 12% threshold for each photo station in both the forests subjected to low-impact and high-impact logging practices.

RESULTS

Fire effects on tree mortality and regeneration

Tree mortality in the burned plot of the logged forest at Fazenda Vitoria was high after the fire. Forty-four percent of the 1010 trees ≥10 cm dbh in the burned

TABLE 2. Tree mortality for trees ≥10 cm in diameter at breast height (dbh) at Fazenda Vitoria study site near Paragominas, Pará, Brazil.

dbh size class (cm)	Burned plot: Trees contacted by fire %	No.	Burned plot: Trees not contacted by fire %	No.	Burned plot: All trees %	No.	Unburned plot, all trees %	No.
10–19.9	59	504	11	133	49	637	3	341
20–29.9	53	163	2	54	40	217	3	134
30–39.9	28	74	5	22	22	96	2	58
40–49.9	58	33	33	6	49	39	0	28
50–59.9	37	8	0	8	19	16	0	8
≥60	67	3	0	2	40	5	0	9
Total all classes	55	785	8	225	44	1010	3	578

plot (2.75 ha) died during the 1.5-yr period following the fire while only 3% of the 578 trees ≥10 cm dbh died in the unburned plot (1.75 ha) (Table 2). However, when only considering the trees contacted by fire (78% of 1010), the total mortality rate for all classes was 55% in the burned plot. Fifty-nine percent of the currently-sawn timber species ≥10 cm dbh that were contacted by fire died. Eight percent of the 225 trees not contacted by fire died (Table 2).

Tree mortality following fire does not necessarily occur rapidly. One month after the fire, we estimated (based on visual inspection of leaf abscission and the cambium layer) that only as many as 38% of the fire-contacted trees would die vs. the 55% recorded 1.5 yr after the fire.

Trees in this region have thin bark (7.1 ± 0.14 mm [$\bar{X}$ ± 1 SE]; Uhl and Kauffman 1990), thus these species are vulnerable to low-intensity ground fires. We found that 42% of 451 trees with only minor fire damage (small surficial scars on a small proportion of the bark),

TABLE 3. Results of ANOVA for the effects of ground (no-slash, slash, skid-trail) and burn (burned, unburned) treatments and census time (1 yr before fire, 15 mo after fire) on density of regenerating plants ≥10 cm tall in four species groups in large gaps 1 yr before fire and 15 mo after fire at Fazenda Vitoria near Paragominas, Pará, Brazil. The whole-plot and split-plot error df were both 13.

Source of variation	df	*F* statistics: Non-timber tree and shrub species	Vine/liana species	Pioneer species	Timber species
Ground, G	2	10**	5.5*	5.7*	2.5
Burn, B	1	0.7	0.2	1.9	0
G × B	2	3	0.8	1.9	0.8
Census C	1	7.3*	7.6*	13.9**	8.2*
C × G	2	2.2	3.6	9.4**	1.3
C × B	1	0.2	0.2	11.3*	2.9
C × G × B	2	2	2.8	12.6**	0.4

* $P < 0.05$; ** $P < 0.005$.

PLATE 1. Burned logging gap one month after the fire in the selectively logged rain forest at the Fazenda Vitoria study site near Paragominas, Pará, Brazil. The area in the foreground shows the remains of logging slash that fueled a locally higher intensity fire, which cleared the layer of regenerating saplings and generally increased fire damage to nearby trees.

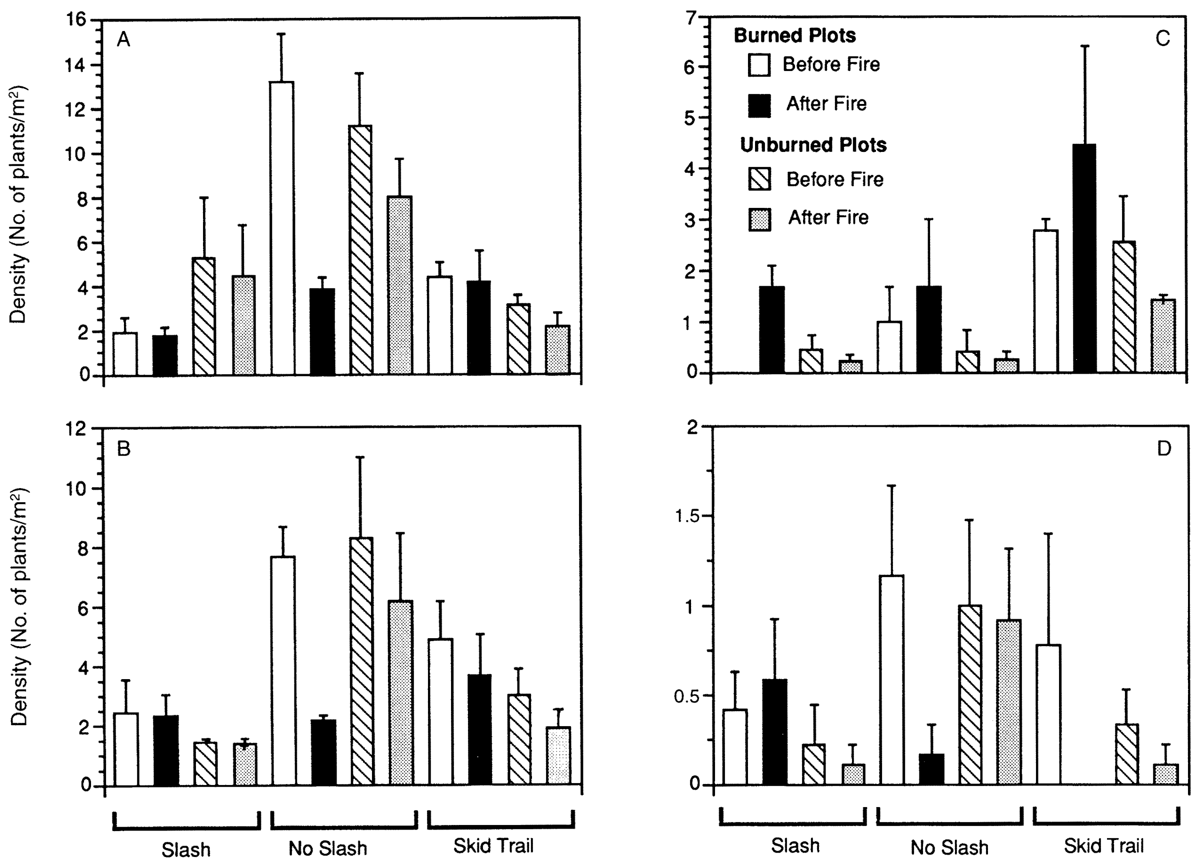

FIG. 3. Density of regenerating plants ≥10 cm tall in burned and unburned slash, no-slash, and skid-trail treatments in large logging gaps 1 yr before fire and 15 mo after fire at Fazenda Vitoria near Paragominas, Pará, Brazil. (A) Non-timber tree and shrub species, (B) vine/liana species, (C) pioneer species, and (D) timber species. Data are means + 1 SE. Note that the *y*-axis scale changes in each panel. Unidentified individuals and forbs (always ≤6% of total density) are excluded.

died following the fire. Percentage mortality increased as fire damage increased; of 114 trees that experienced severe damage (extensive deep scars and split/peeling bark), 84% were dead after 1.5 yr. Seventeen percent of the trees in the burned plot had fallen during the 1.5 yr after the fire.

Changes through time in the density of regenerating plants ≥10 cm in height varied greatly between the treatments, with the most notable response to fire occurring in the five genera of pioneer species (*Cecropia, Banara, Vismia, Solanum,* and *Trema*). Densities of non-timber tree and shrub species and vine/liana species were at their highest before fire (1.9–13.2 plants/m²) and decreased by 5–71% in all burned and unburned treatments after fire (Fig. 3A and B). Between the last pre-fire census and the post-fire census the density of non-timber tree and shrub species and vine/liana species groups was not significantly affected ($P < 0.05$)

TABLE 4. Density (number of plants/m²) of regenerating plants ≥10 cm tall in four species groups in small logging gaps 1 yr before fire and 15 mo after fire at Fazenda Vitoria near Paragominas, Pará, Brazil. Data are means and 95% confidence intervals.

	Non-timber tree and shrub species		Vine/liana species		Pioneer species		Timber species	
Treatment	Pre-fire	Post-fire	Pre-fire	Post-fire	Pre-fire	Post-fire	Pre-fire	Post-fire
Burned slash (n = 4 plots)	2.6 ± 1.7	1.3 ± 2.3	2.1 ± 1.0	1.4 ± 2.1	0	1.7 ± 2.0	0.3 ± 0.8	0.3 ± 0.7
Unburned slash (n = 3 plots)	6.4 ± 11.3	5.0 ± 11.0	2.1 ± 4.8	2.4 ± 3.9	0.1 ± 0.4	0.2 ± 0.5	0.4 ± 0.5	0.4 ± 0.5
Burned no slash (n = 3 plots)	15.4 ± 6.8	4.2 ± 1.0	4.8 ± 3.4	3.9 ± 2.1	0.1 ± 0.4	2.4 ± 6.5	0.2 ± 1.0	0.2 ± 1.2
Unburned no slash (n = 3 plots)	15.0 ± 23.7	10.6 ± 21.6	5.6 ± 6.5	3.2 ± 3.9	0.7 ± 2.8	0.9 ± 3.8	0.4 ± 1.3	0.7 ± 0.8

by fire, but was by ground treatment (slash, no slash, skid trail) and by census time (1 yr before fire, 15 mo after fire) (Table 3).

The density of pioneer species following fire in all burned treatments (1.7–4.4 plants/m²) was at least 60% greater than densities recorded 1 yr before fire. This increase seems to be largely related to the fire itself; between the last pre-fire census and the post-fire census, pioneer species' density increased 60–70% (0.7–1.7 plants/m²) in all burned treatments and decreased 40–50% (0.1–1.3 plants/m²) in all unburned treatments (Fig. 3C). There were significant ($P < 0.05$) effects of ground and census time × burn interaction, and highly significant ($P < 0.005$) effects of census time, census time × ground interaction, and census time × ground × burn interaction (Table 3).

Regenerating timber species generally occurred at lower densities than the other species groups but their future economic value increases the importance of examining their status through time. In the 2 yr prior to the fire the only notable (>10%) changes in density of timber species included a doubling in the unburned no-slash treatment and a 50% decrease in the unburned skid-trail treatment. Between the 1991 and the post-fire censuses there was a significant census-time effect (Table 3); in the post-fire census, all burned treatments had decreased by over 80% (0.8–1.0 plant/m²) except for slash, which increased by 50% (0.2 plant/m²), while unburned slash, no-slash, and skid-trail treatments decreased by 10–70% (0.1–0.2 plant/m²) (Fig. 3D). Overall, there was a 58% (11 plants/m²) decrease in timber species in the burned treatments, more than twice the decrease recorded in the unburned treatments.

Sample sizes were somewhat low (2–4 plots per treatment) and variances relatively high in the large logging-gap data set, but measurements 1 yr before fire and 15 mo after fire in the small logging gaps substantiate the trends in densities found in the large gaps, especially for non-timber tree and shrub species, vine and liana species, and pioneer species (Table 4). The density of non-timber tree and shrub species decreased by 22–73% (1.3–11.2 plants/m²) in all burned and unburned treatments, with the greatest decrease occurring in the burned no-slash treatment. Vine and liana density decreased by 19–43% (0.7–2.4 plants/m²) in all burned and unburned treatments except for a 12% (0.3 plant/m²) increase in unburned slash. As in the large-gap plots, pioneer species' density increased dramatically in the burned treatments. Unlike the large-gap plots, the density of timber species changed negligibly in all small-gap treatments (Table 4).

Fuel dry-down rates in a forest subjected to high-impact logging practices

In the Fazenda Sete forest subjected to high-impact logging practices 8 mo earlier, fuels dried most rapidly in the largest logging gaps that received the most solar insolation. Following 1 cm of rain, fuel sticks in large gaps that received 320–460 min of direct photon flux density (PFD) crossed the 12% fire threshold (the point below which the ignition and spread of fire is possible) after 6 rainless days (Fig. 4A). Areas receiving 160–300 min of direct PFD became fire susceptible after 15 d. However, some areas receiving little solar radiation also became fire susceptible; two areas receiving 48 and 58 min of direct PFD, crossed the 12% fire threshold after 15 rainless days. Sixteen days after rain, fuel sticks in forest patches receiving 0–30 minutes of direct PFD had dried to 15% moisture content (Fig. 4A). The fuel stick in the uncut primary forest dried to 18% moisture content after sixteen rainless days. The window of fire susceptibility was not just between 1300 and 1500, the time of lowest humidity and when measurements were made; 84% of 159 additional measurements taken between 1600 and 1730 were less than or equal to earlier measurements.

In addition to the quantity of direct incoming solar radiation, the dense layer of knee-high regrowth present in some plots influenced fuel dry-down rates. After

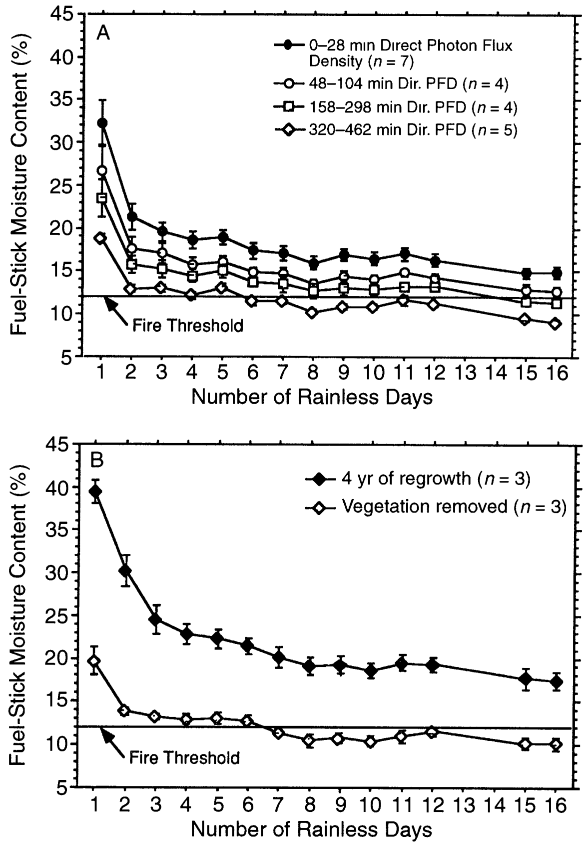

FIG. 4. Fuel dry-down in the logged forest over 16 rainless days following a 1-cm rainstorm at Fazenda Sete near Paragominas, Pará, Brazil. All data are means ± 1 SE; n = number of fuel-stick stations. (A) Forest 8 mo after being subjected to high-impact logging. Measurement stations grouped in classes of minutes of direct photon flux density ($mol \cdot m^{-2} \cdot d^{-1}$). (B) Three large (>700 m²) gaps in a forest 4 yr after being subjected to high-impact logging.

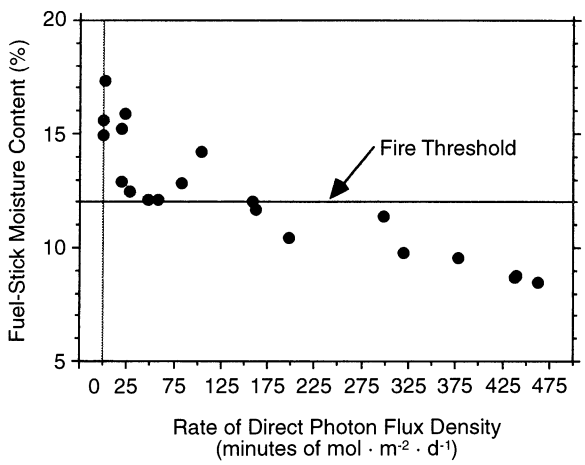

FIG. 5. The relationship between rate of direct photon flux density and fuel-stick moisture content on the sixteenth rainless day in a forest 8 mo after being subjected to high-impact logging at Fazenda Sete near Paragominas, Pará, Brazil (n = 20 fuel-stick stations).

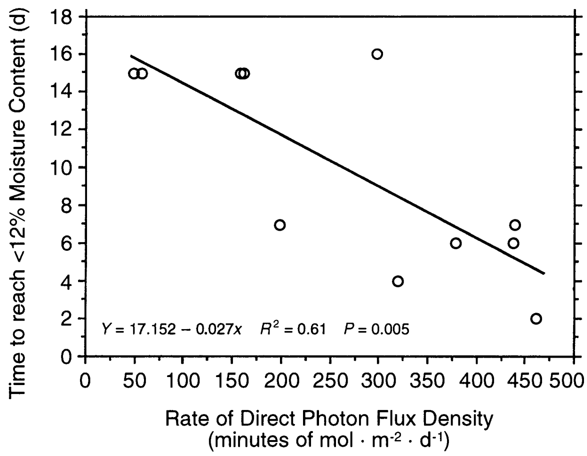

FIG. 6. Linear regression of the rate of direct photon flux density and the number of rainless days required to reduce fuel moisture content below the 12% threshold in the forest logged 8 mo earlier at Fazenda Sete, near Paragominas, Pará, Brazil (n = 11 fuel-stick stations).

five rainless days, the mean moisture content of fuel sticks placed under the regrowth of three large (>700 m^2) gaps remained 2% higher on average than fuel sticks 1 m away in the open. However, by day 16 the mean moisture content of these vegetated stations had dropped below 12%. For two of three paired vegetated and unvegetated stations receiving the most sun, dry-down to below 12% was 8–10 d longer under knee-high vegetation.

Four years of regrowth following logging had a dramatic effect on fuel moisture conditions in three large gaps receiving 340–520 min of direct PFD/d. Clearing the 3–5 m high vegetation cover over a 100-m^2 area caused fuel conditions to mimic those found in the recently logged forest described above; fuel sticks in the clearing dried to <12% moisture content after six rainless days. However, fuel-stick moisture content in the dense regrowth was 17.4% ± 1.8 ($\bar{X}$ ± 1 SD) after 16 rainless days (Fig. 4B).

There is a negative relationship between fuel moisture content and the rate of direct PFD. The 11 stations that dropped below 12% fuel moisture content in the 16 days of measurements received the greatest rates of direct PFD (Fig. 5). For these stations, there is a significant negative relationship between the rate of direct PFD and the number of days it takes to reach the 12% fire threshold (r^2 = 0.61, P = 0.005, n = 11 fuel-stick stations) (Fig. 6), suggesting a model of continuous dry-down. However, the 11 stations reached the 12% level in two groups, one at 2–7 d after rain and the other at 15–16 d after rain, suggesting a more complex relationship than the linear model may imply (Fig. 6).

Fire susceptibility of forests subjected to low-impact vs. high-impact logging practices

The 25-ha forest stand subjected to low-impact logging lacked gaps with high rates of direct PFD even though almost the same number of trees was extracted and the wood volume harvested was actually greater than in the forest subjected to high-impact logging (Fig. 7). The most open photo station in the forest subjected to high-impact logging (n = 58 stations) received 480 min of direct PFD vs. a maximum of only 274 min direct PFD in the forest subjected to low-impact logging (n = 54 stations). In the fuel-stick moisture study, all stations receiving >150 min of direct PFD crossed the fire threshold within 16 rainless days (Fig. 5). Twenty-one percent of the 58 photo stations in the forest subjected to high-impact logging vs. 13% of the 54 stations in the forest subjected to low-impact logging received >150 min of direct PFD/d.

Applying the regression model (Fig. 6) of the relationship between the number of days required to cross the 12% fire threshold and the rate of direct PFD converts the PFD measurement into a "days-to-fire susceptibility" estimate for each photo station, thus providing the opportunity to further compare the fire susceptibility of the forests subjected to high-impact and low-impact logging practices. The equation only applies to the stations that crossed the threshold within 16 d, which in the fuel-stick measurements was all stations receiving >150 min direct PFD/d. Since two stations receiving less solar radiation crossed the threshold, this is a conservative application of the equation.

Photo stations in the forest subjected to high-impact logging began to reach fire susceptibility as soon as 4 d after rain, while no stations in the forest subjected to low-impact logging became fire susceptible until 10 d after rain. Examining rain data over the 10-yr period 1983–1992, 4–9 d rainless periods have occurred twice as often as 10–16 d periods (5.9 ± 2.5 times/yr vs. 3.0 ± 1.7 times/yr, respectively; $\bar{X}$ ± 1 SD). During rainless periods ≥10 d, when areas of both forests subjected to

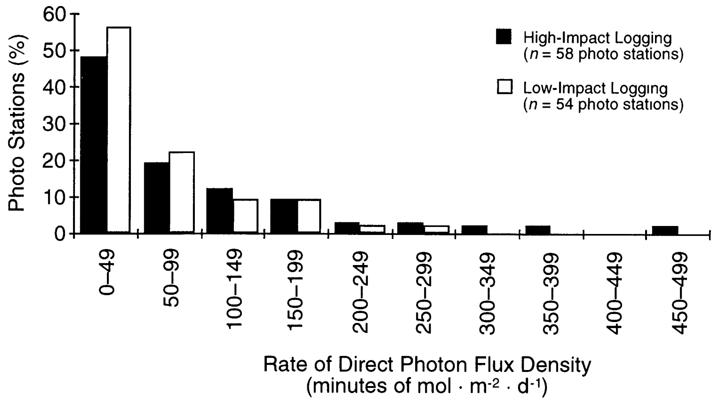

FIG. 7. Percentage of canopy photo stations in each class of photon flux density in a 25-ha forest stand subjected to high-impact logging and a 25-ha forest stand subjected to low-impact logging at Fazenda Sete near Paragominas, Pará, Brazil. The forests subjected to high-impact and low-impact logging had nearly identical tree diameters ($\bar{X} \pm 1$ SE) for trees >25 cm in diameter at breast height before logging (35.0 ± 0.2 cm and 34.9 ± 0.2 cm, respectively) and nearly the same number of harvested trees (94 and 92, respectively).

low-impact and high-impact logging are susceptible to fire, there is a notable difference in the proportion of each forest that can burn; combining the frequency of 10–16 d rainless periods with the rate of dry-down to 12% derived from the regression model, produces a highly significant ($P < 0.0001$), two-fold difference in the proportion of photo stations susceptible to fire in the forest subjected to low-impact logging (8.2 ± 1.0% [$\bar{X} \pm 1$ SE]) vs. those of the forest subjected to high-impact logging (17.0 ± 0.9%) (unpaired t test on logistic-transformed proportions [Bessin et al. 1990]; t = 5.9; n = 29 rainless periods for each forest).

Discussion

Impacts of fire on forest structure and composition

The fire at Fazenda Vitoria seriously altered the structure and composition of the selectively logged forest. It caused a second round of fragmentation of the forest (Fig. 2). First, loggers extracted 3 trees per hectare, severely damaged 121 trees ≥10 cm in diameter at breast height (dbh) per hectare in the logging process, and increased the opening of the canopy by 25%, leaving a heterogeneous mix of high- and low-forest fragments and gaps (Verissimo et al. 1992). Four years later, the accidental fire precipitated the death of 161 trees per hectare. The death and eventual fall of these trees further increased the fragmentation of the high- and low-forest patches, leaving them more susceptible to windthrow and to further alteration of forest structure and composition. Similar patterns of canopy fragmentation were recorded in logged and then burned forests in Malaysia (Woods 1989).

Thick bark and vegetative sprouting could be important determinants of the persistence of individual trees following fire (Kauffman and Uhl 1990). However, sufficiently protective bark properties were not prevalent in the burned forest of Fazenda Vitoria; many trees (42%) that experienced only low levels of bark damage died. Basal and/or epicormic sprouting following a burn has been observed in 59% of surveyed species in this region and can provide a rapid replacement of vegetative cover (Kauffman 1991). While sprouting was occasionally observed at the study site, the timber potential of any such tree is usually low.

The fire at Fazenda Vitoria shifted the trajectory of forest regeneration further towards pioneer species. While the density of non-timber tree and shrub species, vine/liana species, and timber species generally decreased, the density of pioneer species increased by 60–70% to levels higher than that found anytime before the fire in all burned treatments, and decreased 40–50% in all unburned treatments. Kauffman and Uhl (1990) postulated that the high seed-production rates, good long-distance dispersal, persistence in the seed bank, and rapid growth rates of early successional species in high-light environments make them strong contenders in post-fire environments. While the density of pioneer species following logging in the large gaps was greatest in the skid trails, fire also increased the density in slash and no-slash areas (Fig. 3C). The scraping of the ground that accompanies the creation of a skid trail provided a seedbed for pioneer species following logging. Fire served much the same purpose for areas with and without slash by leaving a nutrient-rich ash layer over exposed soil and killing the post-logging regeneration in the gaps, thus converting once low-light environments back into high-light environments.

At least three pioneer genera in this study (*Cecropia* spp., *Vismia* spp., and *Solanum* spp.) include species with an absolute light requirement for germination (Uhl and Clark 1983). In a study in the northwestern Amazon, burning significantly reduced the seed bank from that found in unburned forest. However, post-fire seed dispersal of *Cecropia* spp., *Solanum* spp., and *Vismia* spp. via birds and bats increased these species' abundance (Uhl et al. 1981). While the absolute contribution of the seed bank to seedling establishment in the Fazenda Vitoria burned forest is unknown, the unburned islands of vegetation (Fig. 2B) and the delayed tree mortality probably provided sufficient habitat for birds and bats to disperse seeds of pioneer species into the burned areas.

Woods (1989) recorded an increase in woody creepers following fire in logged forests of Malaysia. Vascular cambium deep inside of the stem tissue of lianas may provide for significant tissue regeneration after mechanical wounding (Dobbins and Fisher 1986) and also protect the liana from fire (Kauffman and Uhl 1990). However, in all regeneration treatments at Fazenda Vitoria there was an overall decrease in vine and liana density following fire. Fire severity and drought may have decreased the effectiveness of usually protective stem morphology; in burned, unlogged primary forests of East Kalimantan, Indonesia, which showed similar post-fire tree mortality to that of Fazenda Vitoria, there was >90% mortality for lianas (Leighton and Wirawan 1986).

In addition to the decrease in vines and lianas, there was an overall 58% decrease in the density of timber species in burned areas of large gaps following fire vs. a 24% overall decrease in unburned areas. Moreover, the death of 59% of the fire-contacted timber trees ≥10 cm dbh has notably reduced the seed source for the replacement of lost timber regeneration. As one proposed management system for the region relies on natural timber regeneration and stocking (Verissimo et al. 1992), we predict that the economic and ecological feasibility of forest management at the Fazenda Vitoria burned, logged forest has been greatly diminished by fire.

Fuel dry-down rates in a forest subjected to high-impact logging practices

Logging gaps were the areas of the most rapid dry-down in the logged forest and thus the focal points for fire susceptibility. Medium-sized gaps (≈200–700 m^2) became fire susceptible after 15–16 d and large gaps (>700 m^2) became fire susceptible after 2–7 rainless days. There was a significant negative correlation between days to cross the 12% fire threshold and rate of direct photon flux density (PFD), thus providing a useful predictor of fire susceptibility during the time when measurements were made. Deviations from this linear model could be due to numerous factors. Brown (1993) found that small gaps exhibit more variability in such microclimatic variables as maximum air temperature and relative air humidity. He suggests that on any given day there is a greater chance that partial cloud cover would block insolation from reaching small gaps than from reaching larger gaps. This could explain why two fuel-stick stations became fire susceptible even though SOLARCALC (Chazdon and Field 1987) estimated that they received at least 63% fewer minutes of direct PFD/d than the other stations that reached fire conditions. Density of undergrowth and/or distance to areas of higher humidity (e.g., forest patches) may also explain deviations from the linear model. The measurement site with the least dry-down relative to the regression line in Fig. 6 was surrounded by an 80–100 cm cover of regrowth *and* was within 15 m of two different forest patches, unlike other stations with only dense regrowth.

Besides the effect that incoming solar radiation has on the rate of fuel dry-down, time since logging also affects incoming solar radiation and, therefore, the rate of fuel dry-down. Eight months of regrowth in a forest subjected to high-impact logging slowed fuel dry-down to the fire susceptibility point by at least 8 d. Four years of regrowth had an even more dramatic effect on dry-down; after 16 rainless days fuel-stick moisture was nearly equal to that in uncut primary forest. Fetcher et al. (1985) found that vapor-pressure deficit, the driving force of fuel dry-down (Pyne 1984), in a 400-m^2 gap was nearly equal to that in primary forest understory after only 2 yr of gap regrowth.

This study was carried out during the second rainless period greater than 6 d following a prolonged wet season. As the dry season progresses, plant-available soil water decreases (Nepstad et al. 1994), probably creating increased water stress for relatively shallow-rooted regeneration, a decrease in transpiration rates, decreased leaf area, and potentially decreased resistance to fire. Indeed, even after 3 yr of regrowth in the Fazenda Vitoria logged forest, 72% of the gap area burned in a late dry-season burn (Fig. 2B). There is a negligible difference in slopes of dry-down rates between the most open areas and the more closed areas (Fig. 4A) suggesting that these closed-canopy forest patches will further approach the threshold, if not cross it, in a longer rainless period. Again, the fire at Fazenda Vitoria supports this possibility; 78% of the high-forest area burned (A. Holdsworth, *unpublished data*).

Forest management and the potential for fire reduction

Fire reduces the potential value of the forest, but by reducing the size of logging gaps low-impact logging techniques can reduce the risk of fire. The low-impact logging at Fazenda Sete reduced the mean gap size by 53% relative to the high-impact logging (Johns et al., *in press*). This reduction in mean gap size resulted in the absence of areas receiving large quantities of direct solar insolation (Fig. 7). The decreased solar insolation reaching the forest floor resulted in a greatly reduced probability of fire in the forest subjected to low-impact logging vs. high-impact logging practices. In addition to the other benefits that low-impact logging brings to the forest owner (e.g., reduced logging waste, lower machine operating costs, and enhanced forest regrowth potential—see Uhl et al. 1997), this study suggests that reduced fire susceptibility is an added benefit.

The extra U.S. $50/ha required to carefully plan and execute logging operations using low-impact logging techniques is reduced by the accumulation of benefits; thus the more benefits that can be accrued, the more economically feasible low-impact logging will be. For the average extraction intensity found in the Paragominas region (30–40 m^3/ha), the added costs of low-impact logging would be completely recovered by the additional benefits (P. Barreto, P. Amaral, E. Vidal, and C. Uhl, *unpublished manuscript*).

While the economic benefit of decreased fire risk has not been quantified, decreased fire risk adds an element of insurance for the forest owner. As more loggers own their own forests, there is an increasing concern over the threat of fire to their forests. Along with sustainably cutting their forests every 30–50 yr with low-impact logging techniques, forest owners must be assured that their long-term investment in their forests will not be threatened by accidental fire. This study has demonstrated the reduced fuel dry-down rates in forests subjected to low-impact vs. high-impact logging practices. With the reduction in logging wastes, fuel loads have also been effectively reduced by a still unknown but notable amount. The combination of reduced fuel dry-down rates and reduced fuel loads can effectively provide forest owners with assurance that their forests are more protected from fire than if they pursued typical logging practices.

The decreased fire susceptibility that we observed in the forest subjected to low-impact logging does not mean that fire is not a concern. Selectively logged forests are typically adjacent to fire-prone pastures. In the most intense dry-seasons (El Niño years), even a forest subjected to low-impact logging is likely to be at risk to fire if it is near pasture or slash-and-burn fields. Besides the reduced flammability that low-impact logging can bring, protection in the form of fire breaks adjacent to fire-prone areas, such as pasture, is required. Goldammer (1988) says that fire management in tropical lands must focus on fuel management, such as the creation of fuel breaks. The requirement of a fuel/fire break around all selectively logged forests is a component of one potential forestry code for the Brazilian Amazon. This code could be called "edict 5/50/5" and would limit logging to 5 trees/ha, require 50 yr between logging cycles, and require a 5-m-wide swath of ground cleared of fuels (Uhl et al. 1997). This 5-m fuel break surrounding each tract of selectively logged forest would protect the forest as long as it is scrupulously maintained. An alternative would be to preserve a 1-km-wide swath of intact primary forest surrounding selectively logged areas. These borders of primary forest could serve as fire breaks if they are tapping deep water reserves (Nepstad et al. 1994) and would thus be able to maintain fuel moisture levels well above the fire threshold. Fire breaks of primary forest have two distinct advantages over a 5-m-wide fuel break. First, there is no need for the frequent maintenance that is critical for an effective fuel break. Second, 1-km-wide primary forest fire breaks enhance ecosystem function; they facilitate regeneration of the selectively logged areas by providing seed sources of primary forest species, corridors for passage of seed dispersers, and potential sources of non-timber forest products.

To further the potential for forest fire reduction in the increasingly fragmented eastern Amazonian landscape, we need to study fire risk and its economic costs in selectively logged forests subjected to a range of logging intensities. These studies should include the driest periods of multiple dry seasons in order to bolster our understanding of the relationship among fire risk, intensity of logging, and time since logging. Finally, we need studies on the fire resistance and economic costs of fire breaks of various widths through many climatic conditions. The results of these studies could further demonstrate that increased fire resistance is another reason to practice low-impact logging techniques and that the strategic preservation of intact primary forest as natural fire breaks may be a cost-effective way to reduce the economic and ecological damage wrought by fire on selectively logged forests.

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