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Abrupt increases in Amazonian tree mortality due to drought–fire interactions

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Interactions between climate and land-use change may drive widespread degradation of Amazonian forests. High-intensity fires associated with extreme weather events could accelerate this degradation by abruptly increasing tree mortality, but this process remains poorly understood. Here we present, to our knowledge, the first field-based evidence of a tipping point in Amazon forests due to altered fire regimes. Based on results of a large-scale, long-term experiment with annual and triennial burn regimes (B1*yr* and B3*yr*, respectively) in the Amazon, we found abrupt increases in fire-induced tree mortality (226 and 462%) during a severe drought event, when fuel loads and air temperatures were substantially higher and relative humidity was lower than long-term averages. This threshold mortality response had a cascading effect, causing sharp declines in canopy cover (23 and 31%) and aboveground live biomass (12 and 30%) and favoring widespread invasion by flammable grasses across the forest edge area (80 and 63%), where fires were most intense (e.g., 220 and 820 kW·m^{-1}). During the droughts of 2007 and 2010, regional forest fires burned 12 and 5% of southeastern Amazon forests, respectively, compared with <1% in nondrought years. These results show that a few extreme drought events, coupled with forest fragmentation and anthropogenic ignition sources, are already causing widespread fire-induced tree mortality and forest degradation across southeastern Amazon forests. Future projections of vegetation responses to climate change across drier portions of the Amazon require more than simulation of global climate forcing alone and must also include interactions of extreme weather events, fire, and land-use change.

forest dieback | fireline intensity | stable states | MODIS | fire mapping

Large areas of moist tropical forests are being altered by land-use practices and severe weather. People are clearing, thinning, and changing the composition of tropical forests (1, 2). Severe drought events superimposed upon these land-use activities increase forest susceptibility to fires (1–5). In the 2000s, for example, 15,000–26,000 km^2 of Amazonian forests burned during years of severe drought (6). Widespread forest fires may become even more common in the Amazon Basin if the frequency of extreme weather events increases, particularly in the southeastern Amazon (1, 7). However, most model simulations of future trajectories of Amazonian forests have relied on global and regional climate forcing that do not consider the effects of fire on vegetation dynamics and structure (8–10).

Our ability to predict future fire regimes in moist tropical forests is constrained by a lack of understanding of what triggers and controls high-intensity fires (7, 11). In nondrought years, primary forests typically do not catch fire during the dry season because the fine fuel layer is too humid to carry a fire (12). This characteristic of primary forests helps explain why forest fires were less frequent in pre-Colombian times than today (13), although indigenous peoples of the Amazon have used fire as a management tool for hundreds or thousands of years (14). Current anthropogenic disturbances in moist tropical forests (e.g., logging, forest conversion for crops and livestock, and the resulting fragmentation of forests) tend to thin forest canopies (5, 11) and expose forest interiors to warm air flowing horizontally from neighboring clearings, allowing the forest floor to dry more rapidly during rainless periods. When forest fires do occur under average weather conditions, they typically move through the understories slowly (15–25 m·hour^{-1}), release little energy (50 kW·m^{-1}), and are of short duration (4, 5, 15), extinguishing at night when relative humidity increases. Despite their low intensity, understory fires still exert strong influences on forest dynamics and structure because many tropical tree species are thin-barked and vulnerable to fire damage (12, 16, 17).

During years of severe drought, Amazon forest fires are atypically intense, killing up to 64% of the trees (18, 19). This happens because fuel (e.g., twigs, leaves, branches, etc.) not only becomes drier, but also tends to become more abundant due to drought-related leaf and branch fall (20). Thus, compared with low-intensity fires that occur in nondrought years, severe droughts can trigger high-intensity fires that kill more trees. Unfortunately, the role of extreme weather events in the fire dynamics of moist tropical

Significance

Climate change alone is unlikely to drive severe tropical forest degradation in the next few decades, but an alternative process associated with severe weather and forest fires is already operating in southeastern Amazonia. Recent droughts caused greatly elevated fire-induced tree mortality in a fire experiment and widespread regional forest fires that burned 5–12% of southeastern Amazon forests. These results suggest that feedbacks between fires and extreme climatic conditions could increase the likelihood of an Amazon forest "dieback" in the near-term. To secure the integrity of seasonally dry Amazon forests, efforts to end deforestation must be accompanied by initiatives that reduce the accidental spread of land management fires into neighboring forest reserves and effectively suppress forest fires when they start.

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Fig. 1. High-resolution image (i.e., 1.85 m) of the experimental area in 2011 captured with the sensor Worldview-2. The dashed line represents the border between the North–South forest edge (0–100 m) and the forest interior (100–1,000 m). The North–South edge of the plots is bordered by a road and open agricultural fields, and the other plot boundaries are in contiguous forest. The control represents an unburned area, and B1*yr* and B3*yr* areas that were burned annually and every 3 y, respectively, from 2004 to 2010 (with the exception of 2008).

forests is difficult to study because they are hard to predict. As a result, the relationships between fire-induced tree mortality and extreme weather remain poorly understood, restricted mostly to postfire observations of tree mortality.

To fill this gap, in 2004 we established a large-scale, long-term prescribed forest fire experiment in a transitional forest (between Amazon forests and savannas) in the southeastern Amazon (Figs. 1 and 2)—a region that is highly vulnerable to changes in fire regime, climate change, and their interactions (2). The experimental area consists of three adjacent 50-ha (1.0 × 0.5 km; Fig. 1) plots burned annually (B1*yr*), every 3 y (B3*yr*), or not at all (control) to represent a range of possible future forest fire frequencies (details in ref. 21). We used within-plot variability between the forest edge (0–100 m into the forest from the adjacent agricultural area) (Fig. 1) and forest interior (100–1,000 m) and the temporal variability in weather between 2004 and 2010 to address two questions: (*i*) Are there weather- and fuel-related thresholds in fire behavior that are associated with high levels of fire-induced tree mortality across two different fire regimes? (*ii*) What are the effects of an intense fire event on forest structure, flammability, and aboveground live carbon stocks? We also conducted a regional analysis of weather and fire scars to assess the spatial-temporal dynamics of forest fires in the 87,000 km^2 of remaining forests in the Upper Xingu River Basin (Fig. 1).

Results and Discussion

The 2007 Drought. Precipitation across the Xingu region was lower in 2007 than in any other year during the 2000–2010 period. For example, the maximum climatological water deficit (MCWD), a measure of cumulative water stress (22), averaged −483 mm in 2007 (Fig. 2), representing values that were 20% lower than the average MCWD in the 2000s. These low MCWD values were observed mostly between August and September (~91%), when most Amazon forest fires occur (9, 10, 23). In 2007, ~72% of the Xingu region experienced below-average rainfall anomalies according to the Tropical Rainfall Measuring Mission (TRMM), which indicates the regional nature of this drought.

Our large-scale fire experiment is the only one in neotropical forests to have experienced a severe drought event, coupled with increased temperatures, during the prescribed burns. In 2007, cumulative precipitation was lower than in other years (Fig. 2); the daily relative humidity was 25% lower and the maximum air temperature was 3.6 °C higher than the 7-y dry-season average (Fig. 3); understory air dryness, represented by vapor pressure deficit (VPD), was substantially higher in B3*yr* [95% bootstrap confidence intervals (CI): 3.2, 3.6 kPa] and B1*yr* (CI 3.7, 4.0 kPa) than in other fire years (CI 2.6, 2.7 kPa) (Fig. S1).

Drivers of Fire Intensity. During the experimental fires of 2007, fuel characteristics favored the occurrence of high-intensity fires. Litter moisture content (LMC) in the burned plots was low (9–13%), whereas fine fuel loads generally exceeded the long-term average. For example, leaf litter and 1-h fuel (i.e., woody fuels <0.6 cm in diameter) loads were 30 and 55% greater, respectively, in 2007 than in other years (Fig. 4; $P < 0.01$). In 2007, 1- and 10-h fuels (i.e., woody fuels 2.5–7.6 cm in diameter) were more abundant in B3*yr* than in B1*yr* (i.e., along the forest edge), whereas other fuel-size classes and leaf litter were similar (Table S1).

Fuel load typically correlates positively with fire intensity, but only if the fuel is consumed and its energy is released (24). Here, we present another proxy of fire intensity that accounts for fuel combustion: frontal fire intensity (I), calculated as the product of fire spread rate (r); net heat of combustion (H, kept constant for both plots); and the weight of fuel consumed by the fires (w) ($I = Hwr$) (25). In 2007, frontal fire intensity was (*i*) higher than in 2004 in both plots and (*ii*) higher in B3*yr* (edge: 820 kW·m^{-1}; forest: 319 kW·m^{-1}) than in B1*yr* (edge: 220 kW·m^{-1}; forest:

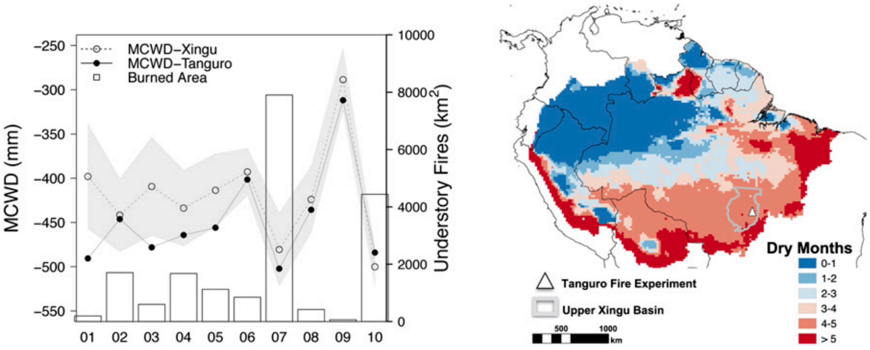

Fig. 2. (*Left*) Annual MCWD between 2000 and 2010 for the Upper Xingu Basin (solid circles) and the experimental field site (Fazenda Tanguro, solid triangles). The shaded area represents the SD of the mean and accounts for the spatial variability in MCWD across the Upper Xingu Basin. (*Right*) Average dry-season length (i.e., number of months with precipitation ≤100 mm) and the locations of both the Upper Xingu Basin (in gray) and the fire experiment (triangle). MCWD and monthly precipitation were derived from the TRMM.

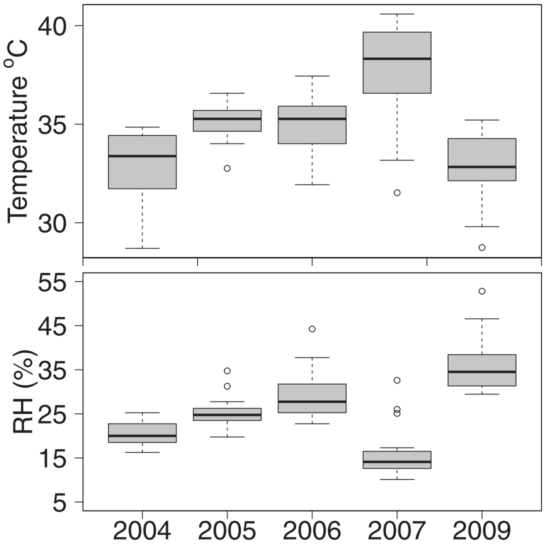

Fig. 3. Air temperatures (*Upper*) and relative humidities (*Lower*) during the experimental fires between 1000 and 1600 hours. These two variables were measured at a meteorological station 4 km from the experimental area.

141 kW·m^{-1}). During the 2007 drought, the differences in fire intensity between treatments were associated mostly with r, given that fuel consumption was similar between plots (Table 1). Also noteworthy is that, whereas the fires self-extinguished at night in all previous nondrought years, this was not the case in B3*yr* during the 2007 drought event, probably due to high VPD, low LMC, and high fuel loads.

Tree Mortality. The most striking result of our experiment was the spike in tree mortality in 2007 (Fig. 5). Compared with the other years, overall fire-induced tree mortality in 2007 was 462% (in B3*yr*) and 219% (in B1*yr*) higher. Increases in the 2007–2008 mortality rates were most pronounced along forest edges (Fig. 5 and Fig. S2), where annualized mortality rates for small trees reached 84% (B1*yr*) and 33% (B3*yr*). By 2009, 50% (B1*yr*) and 60% (B3*yr*) of the trees initially sampled were dead in the burned plots versus 20% in the control (Fig. 6). Although increased mortality was observed in both fire treatments, it was more pronounced along the forest edge of the treatment burned every 3 y, which corresponds to the most common forest fire-return interval in this region (6).

The high mortality rates following the 2007 fires contrasted remarkably with those following the fires in nondrought years. From 2004 to 2006, for example, the experimental fires caused only modest increases in tree mortality compared with the control, and primarily influenced small individuals (Fig. 5). This result suggests that this transitional forest is more resistant to low-intensity fires than wetter Amazonian forests, where postfire mortality rates under drought and nondrought conditions tend to be considerably higher than the ones observed in this study (27, 28).

Mortality rates in the control plot were highest during the 2007–2008 drought, when they averaged 5.0% yr^{-1}. This increased mortality suggests that the 2007 drought was severe enough to alter forest dynamics and structure even in the absence of fire. However, mortality rates in this transitional forest (2–4% yr^{-1}) are generally higher than in wetter forests (1–2% yr^{-1}) (29), making it unclear whether the high 2007–2008 mortality rates in the control plot were a response to drought-related effects.

Drivers of Fire Intensity and Fire-Induced Tree Mortality. The differences in fire-induced tree mortality among years and between fire treatments provide important insights into the drivers of the intense fires of 2007. First, 2007 was drier and warmer than other fire years, and fine fuel loads (along with fuel combustion) were substantially higher. Thus, dry and warm weather conditions and increased fuel loads are the likely causes of the increase in fire intensity in 2007. Second, LMC in 2007 was similar between burn plots, whereas both 1-h fuel loads and mortality rates were higher in B3*yr* than in B1*yr* (particularly along the forest edge). These results suggest that the differences in fire-induced tree mortality between fire treatments were, in part, due to fine fuel loads. High loads of 10-h fuels along the edge of B3*yr* in 2007 also probably contributed to the increased fire intensity and associated increases in tree mortality (e.g., mortality rates were highest along the edge of B3*yr*).

Based on our fire intensity metric ($I = Hwr$), fine fuel loads in 2007 were expected to affect I via their effects on fire spread rates (r), given that the net heat of combustion (H) was constant, fuel consumption (w) was similar between plots, and r was higher in B3*yr* than in B1*yr* (Table 1). One potential explanation for this increase in r in 2007 is that high fine fuel loads in B3*yr* were associated with increased fuel continuity, which in turn increased fire spread rates relative to B1*yr*. These results reinforce the notion that high fine fuel loads (1-h fuels) can cause substantial increases in fire intensity (21). Note that changes in wind patterns in 2007 relative to other years could also partially account for the increases in fire intensity (e.g., by increasing r), but not for the differences between treatments, given that the two burn plots were likely equally exposed to winds in 2007.

Whereas fire intensity explained most of the spatial-temporal patterns of fire-induced tree mortality (Fig. 7), forest tree resistance to fire probably influenced those patterns as well. The fires between 2004 and 2006, for example, may have killed many of the fire-sensitive trees in B3*yr* and B1*yr*, resulting in a surviving community that could be more fire-tolerant (28, 30). Alternatively, individuals that survived but were damaged by the pre-2007 fires may have been more vulnerable when burned again in 2007. Forest community fire resistance and resilience is an

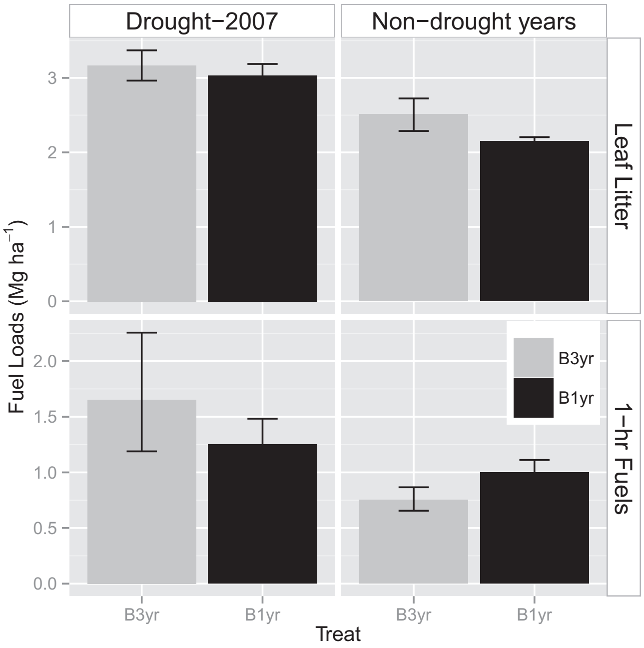

Fig. 4. Fine fuel loads measured in 2007 (*Left*) and in other fire years (2004, 2005, 2006) (*Right*). Litter represents only leaves, whereas 1-h fuels represent twigs ≤0.6 cm in diameter. These fuels were measured in the experimental area using the Brown's planar transect technique (43) along 27 transects per plot.

Table 1. Metrics of fire intensity during the 2004 and 2007 fires in the edge and forest interior of the plots burned every 3 y (B3*yr*) and 1 y (B1*yr*)

		Fuel consumed (kg·m^{-2})		Fire spread rate (m·min^{-1})		Frontal fire intensity (kW·m^{-1})	
Location	Treatment	2004	2007	2004	2007	2004	2007
Edge	B3*yr*	0.95 (1.23)	2.72 (2.05)	0.26 (0.27)	1.01 (0.19)	74.3	819.9
Edge	B1*yr*	1.68 (1.25)	2.12 (1.17)	0.14 (0.99)	0.34 (0.20)	71.1	219.5
Forest	B3*yr*	0.71 (1.18)	1.58 (1.12)	0.26 (0.13)	0.67 (0.14)	55.9	319.1
Forest	B1*yr*	0.50 (0.96)	1.54 (2.88)	0.13 (0.13)	0.31 (0.10)	20.0	141.6

In parentheses, we present 1 SD of the mean.

important topic that should be further investigated across the Amazon Basin, especially for intersite comparisons of forest responses to fire.

In summary, the spike in tree mortality during the fires of 2007 likely resulted from high loads of fine fuel, anomalously dry and hot microclimatic conditions (VPD: >3.2 kPa), and low fuel moisture content (<13%). Together, these conditions appear to have surpassed a threshold of fire intensity beyond which tree mortality increased sharply (Fig. 7).

Live Biomass and Forest Flammability. In response to elevated fire-induced tree mortality in 2007, aboveground live biomass in the burned plots decreased by 12–30% from 2007 to 2008, whereas it remained constant in the control (Fig. 6). These high levels of tree mortality also reduced the leaf area index (LAI), which permits more solar radiation to penetrate the canopy, thus increasing understory air dryness. For example, between 2007 (prefire) and 2008, LAI dropped 23–30% in the interiors of the two burned plots (Fig. 6 and Figs. S3–S5). As a result, dry-season understory VPD (from 0800 to 1800 hours) after the 2007 fires was 45% higher in the burned plots than in the control (Fig. 6 and Figs. S6 and S7).

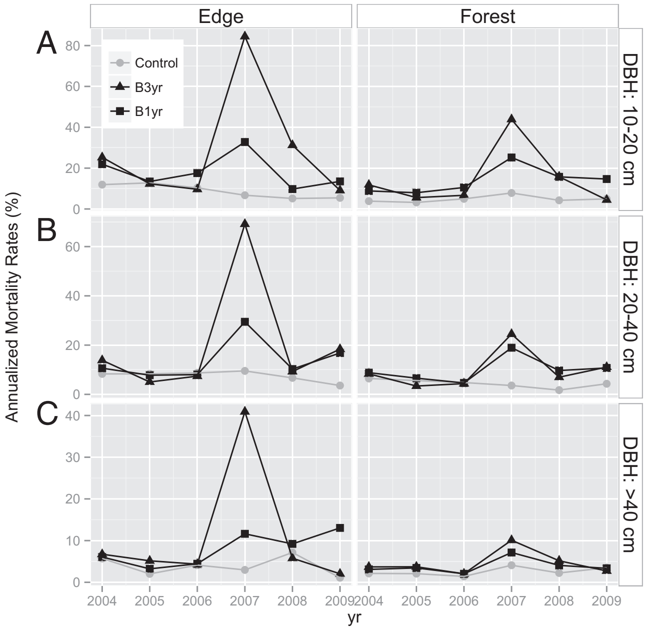

Fig. 5. Annualized tree mortality rates for 2004–2010 in the edge zone and forest interiors for three stem diameter (dbh) size classes: (*A*) 10–20 cm, (*B*) 20–40 cm, and (*C*) ≥40 cm. B1*yr* was burned in 2004, 2005, 2006, 2007, and 2009, and B3*yr* was burned in 2004 and 2007. Mortality rates were calculated using methods described in Balch et al. (26). In 2008 we did not conduct the experimental fires.

Along the edges of the burned plots, grasses invaded in response to low LAI and high grass seed availability (31, 32). Whereas grasses advanced slowly into the burned forest from 2004 to 2006 (1–2 m·yr^{-1}), grass invasion increased substantially after the 2007 fires (e.g., 13–20 m·year^{-1} average from 2008 to 2011), likely in response to fire-related reductions in LAI and associated increases in solar radiation (27, 33). Grass colonization caused a fourfold increase in fire intensity, as represented by r measured during experimental fires in 2010 (33), because grasses accumulated more fine fuel close to the ground surface than the trees that they replaced (31). By 2012, grasses had invaded 8–9.25% of the burned plots, but only 0.06% of the control plot (Fig. S8).

Regional Fire Regime. Insights from our fire experiment indicate that human-driven alteration of the Xingu landscape has already substantially modified the forest fire regime in this region and increased fire-related forest degradation. Specifically, our fire experiment showed high rates of fire-induced tree mortality, particularly along forest edges that experienced previous disturbance and during warm and dry weather. Regionally, deforestation and previous disturbance (by fire or logging) influence these predictors of mortality in three ways. First, by reducing canopy cover and evapotranspiration, deforestation increases average dry-season land-surface temperatures (Fig. S9), which in turn promotes air movement between open fields and neighboring forests. Consequently, fuels along forest edges are expected to become drier, leading to increased fire intensity (34). Second, deforestation fragments the landscape, creating a greater perimeter of forest edges (35). Third, tree mortality associated with previous logging, fire, severe drought, or edge effects can contribute to coarse fuel loads for multiple years as the twigs and branches of standing dead trees gradually decay and fall to the ground. Between 1997 and 2011, the length of forest edges in the Upper Xingu region increased by 34% (Table S2). By 2011, ∼8% of the region's forests were <100 m from a clearing. These deforestation-driven increases in forest edges (35) and regional temperatures are likely to act synergistically to increase the likelihood of high-intensity fires throughout much of the Xingu region.

In addition to promoting high-intensity fires, the regional droughts of 2007 and 2010 created favorable climatic conditions for widespread fires in the southeastern Amazon. In 2007, for example, 12% (7,904 km^2) of the Xingu's forests burned, compared with an average of 0.84% in the nondrought years between 2000 and 2009 (Fig. 2 and Fig. S10). Within the Xingu Indigenous Park, where there are fewer sources of ignition than outside the park (Table S1), nearly 10% of forests burned in 2007. In 2010, the Amazon experienced another drought (36), with widespread understory forest fires that affected 5.4% of the Xingu region (Fig. S10). These extensive fires occurred even as deforestation declined (Table S2), suggesting that weather may have overwhelmed the expected inhibitory effect (on forest fires) of reducing the fire ignition sources that often accompany deforestation.

One insight from our field experiment may explain the increase in burned forest areas across the Upper Xingu region during these

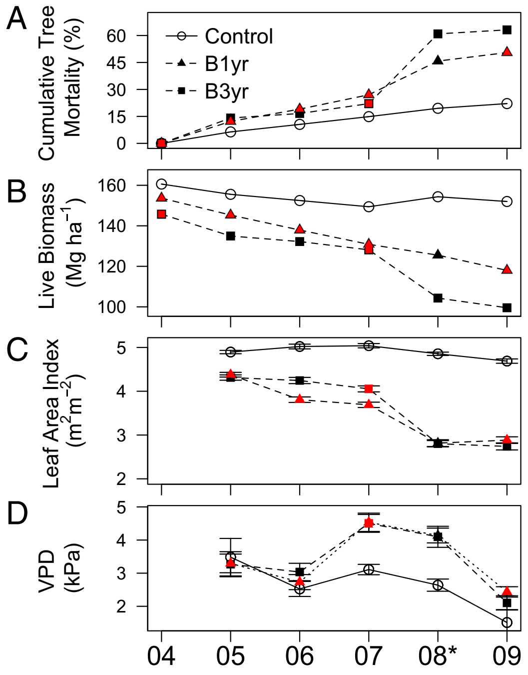

Fig. 6. Temporal patterns in (*A*) cumulative tree mortality for trees ≥10 cm dbh, (*B*) aboveground standing live biomass, (*C*) LAI, and (*D*) forest understory VPD. Symbols in red denote when a given plot was experimentally burned (B1*yr*: 2004, 2005, 2006, 2009; B3*yr*: 2004 and 2007). Note that these values refer to postfire measurements within a given year. In 2008 we did not conduct the experimental fires (*).

drought events. The experimental fires of 2007 did not extinguish at night as they did in nondrought years. If this phenomenon were widespread throughout the region, it would help explain the increases in forest area burned in 2007 and 2010.

Broader Implications. ***Regional Amazon fire regimes.*** Extreme regional weather events and associated increases in fire-related tree mortality are already causing regional forest degradation across the Xingu River Basin, where the seasonally dry climate resembles a substantial portion (~39%) of the Amazon Basin (Fig. 2). These results provide, to our knowledge, the first experimental evidence of the link among extreme weather events, widespread and high-intensity fires, and associated abrupt changes in forest structure, dynamics, and composition. This mechanism of rapid forest degradation could operate over a larger geographical area, such as the "arc of deforestation," where droughts (36), forest fragmentation (20), and forest fires (4, 6) are already common. Understory Amazon fires strongly influence forests located in areas with prolonged dry seasons (21), but more humid forests could also become flammable and susceptible to fire-related degradation (15) as climate and land use change.

Controversy in the literature about a potential Amazon forest "dieback" has relied mostly on models using climate forcing alone (1, 8–10). However, our findings suggest that the interaction between fires and droughts is perhaps a more direct mechanism of abrupt forest degradation for the southeastern Amazon. The future extent, intensity, and severity of forest fires in both drier and wetter parts of the Amazon will depend on the intensity and frequency of droughts and heat waves, the availability of fire ignition sources, and the degree of forest degradation and fragmentation (5). Our results underscore the need for the representation of (*i*) drought events and heat waves in climate models, (*ii*) human-related fire regimes in ecosystem models, and (*iii*) forest fragmentation in scenarios of future deforestation—key factors for understanding future Amazon fire regimes and the trajectory of Amazon forests.

Ecosystem state changes. The observed grass invasion along the forest edges of the experimentally burned plots suggests that high-intensity fires could promote abrupt fire-mediated transitions from forests to new stable states (Fig. S11). Our findings indicate that these transitions are more likely to occur in areas where forests are fragmented, disturbances are frequent, and dry seasons are prolonged (≥4–5 mo) (37–39) (Fig. 2). The future trajectory of Amazonian forests that experience severe weather and forest fires will depend, in part, upon tree species composition (e.g., prevalence of fire-tolerant and resprouting species) and proximity to seed sources of invasive grasses from pastures and savannas. The long-term trajectory of burned Amazon forests is still uncertain, particularly the pervasiveness and persistence of alternate vegetation states.

Landscape management. To secure the integrity of Amazonian forests along the arc of deforestation, efforts to end deforestation in the Amazon must be accompanied by programs and policies that reduce the accidental spread of land management fires into neighboring forests and effectively control forest fires when started. Both of these changes in regional land management seem feasible (3, 35), and several promising initiatives to reduce the probability of forest fires are already underway. Examples of recent efforts include (*i*) development of an early warning system to forecast the locations and intensities of fires (3); (*ii*) implementation of Brazilian federal and state policies to prevent and control forest fires (40); and (*iii*) creation of volunteer fire brigades to fight fires within private farms and indigenous reserves (14, 35, 40). Over a longer time horizon, the future of many forests in the region will require successful mitigation of greenhouse gas emissions to reduce the likelihood of extreme weather events.

Materials and Methods

Fire Experiment. The fire experiment was established in 2004 in the driest portion of the Amazon Basin (Fig. 1) (13° 04′S, 52° 23′N) where the dry season lasts for 4–5 mo, the annual precipitation averages 1,770 mm (2005–2011), and monthly rainfall from May to August is typically below 10 mm (41) (Fig. 2). The dry climatic conditions at this site allowed us to conduct experimental burns even in nondrought years.

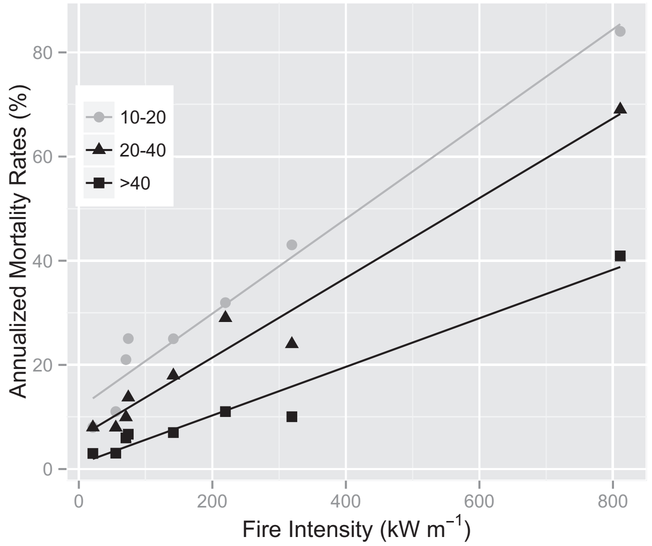

Fig. 7. Relationships between annualized tree mortality rates and fire intensity for three classes of diameter at breast height: 10–20 cm; 20–40 cm; and ≥40 cm. Each point represents an average for the forest interior or edge of B1*yr* or B3*yr*. These data were available for 2004 and 2007.

The experiment was located in an area with no signs of recent fires that was composed of three 50-ha plots: an unburned control, a plot that was experimentally burned every three years (2004 and 2007; B3*yr*), and a plot that was experimentally burned annually from 2004 to 2009 (B1*yr*) with the exception of 2008. Within each 50-ha plot we (*i*) conducted yearly mortality ("top-kill") censuses of trees ≥10 cm in diameter at breast height (dbh) (∼3,000 individuals per plot); (*ii*) mapped trees that entered the 10 cm dbh size-class in 2008, 2009, and 2010; (*iii*) estimated pre- and postfire LAI each year of the study (200 sites per plot; dry and wet seasons) using two LiCor 2000 Plant Canopy Analyzers; (*iv*) monitored hourly VPD in the forest understory (25 cm from the ground) using Onset Hobo U23 Pro v2 Temperature/Relative Humidity data loggers ($n = 45$); (*v*) measured pre- and postfire (± wk) leaf litter and 1-h (0–0.6 cm in diameter), 10-h (0.6–7.6 cm diameter) and 100-h (≥7.6 cm in diameter) fuel loads annually across the experimental area [based on Brown's planar intercept method (42); $n \sim 27$ samples per plot]; and (*vi*) estimated fire spread rate at ∼200 points across the experimental area by measuring fireline movement over time. Litter moisture content (i.e., weight of water per unit dry weight) was estimated from measurements of fine litter collected within circular plots (40 cm in diameter; $n = 90$) 5 min before ignition of the experimental fires. Details of these measurements can be found in Balch et al. (21, 26). Frontal fire intensity was calculated as the product of net heat of combustion, weight of fuel consumed, and rate of spread (25). Net heat of combustion was assumed to be 18,000 $\text{kJ}\cdot\text{m}^{-1}$ for both plots, following Alexander (25). In addition to frontal fire intensity, we measured char heights on all sampled trees as a proxy for intensity because char height typically correlates positively with tree mortality (28). Both char height and frontal fire intensity showed similar patterns as fire intensity, so we present only frontal fire intensity.

Regional Analyses of Weather and Fire Scars. Burn scars in forested areas were mapped for the 127,000 km^2 of the Upper Xingu Region from 2000 to 2010 using MODIS images [details in Morton et al. (43)]. Regional maps of dryness (i.e., MCWD) for 2000–2010 were derived from the TRMM data product for the entire Upper Xingu Region based on methods described by Aragão et al. (22). Maps of land-surface temperature for the dry seasons of 2007 and 2010 (July–September) were derived from the MODIS temperature and emissivity product (MOD11A2). The metrics for landscape fragmentation (edge length and area and deforestation) were calculated for the Xingu Region for 1997 and 2000–2011 based on Landsat-5 TM images (Instituto Nacional de Pesquisas Espaciais); all calculations were performed in the Dinamica-EGO modeling environment.

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