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Amazonian forest resilience inferred from fire-induced changes in carbon stocks and tree diversity

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Supplementary material for this article is available online

Abstract

Understanding the resilience of tropical forests to fire is essential for evaluating their dynamics under climate change and increasing land-use pressures. Here, we assess how different fire frequencies and intensities influence tree mortality and carbon dynamics in southeastern Amazonia. Using a replicated randomized block design with 24 plots (40 × 40 m), we applied four treatments: unburned control, one burn in 2016 (B1), two burns in 2013 and 2016 (B2), and two burns with added fuel (B2+) to increase fire intensity. Forest inventories conducted from 2012 to 2024 measured tree mortality, diversity, composition, and aboveground biomass. Fire frequency and intensity significantly increased mortality, particularly among small trees, but impacts on forest structure and productivity were more nuanced. Aboveground biomass declined modestly in burned plots, with the greatest loss in B2+ (13%). Aboveground net primary productivity dropped immediately post-burn, especially in B2 and B2+, and partially recovered by 2022–2024. In contrast, leaf area index and litterfall rebounded within a couple of years, suggesting a degree of structural and functional resilience. Species richness and composition remained relatively stable in the years following the first experimental fires, but gradually declined and shifted in B2 and B2+ plots beginning in 2014. These results indicate that the experimental forests' resilience to low-intensity and infrequent fires can prevent widespread forest collapse, but repeated and intensified burns likely undermine long-term resilience by altering forest structure, composition, and carbon dynamics. With the southeastern Amazon forests projected to burn more often in the coming decades, our results highlight both the vulnerability and recovery potential of these ecosystems. Maintaining ecological integrity and minimizing additional disturbances that influence fuel availability will be critical for sustaining forest functions under future fire regimes.

1. Introduction

The resilience of tropical forests to disturbances such as fire is critical for understanding the future trajectory of these ecosystems amid climate change and increasing land-use pressures. Amazonian forests have shown some resistance to single fires (Brando *et al* 2016); however, compounding disturbances such as land-use change, fragmentation, and climate warming can erode this resistance (Nobre *et al* 2016, Laurance *et al* 2018, Lapola *et al* 2023, Flores *et al* 2024) by increasing forest dryness and combustible material, leading to more frequent and intense recurrent fires (Cochrane and Laurance 2002). This emerging scenario puts the recovery capacity of these forests at risk and could change plant diversity and species composition, with direct impacts on the carbon cycle (Brando *et al* 2014, 2019, Rappaport *et al* 2018, Esquivel-Muelbert *et al* 2019). Therefore, quantifying the impacts of fire on tree diversity and aboveground carbon stocks under the extreme conditions observed in many agricultural frontiers is particularly valuable (Silva *et al* 2020).

Fires in Amazonian rainforests can have major impacts on aboveground carbon stocks, especially under extreme drought conditions (Gatti *et al* 2014, 2021, Brando *et al* 2016, Rappaport *et al* 2018). Trees are the main carbon reservoirs in these ecosystems, and the mortality induced by fires releases large amounts of carbon into the atmosphere (Barlow *et al* 2003). The resulting reduction in aboveground carbon stocks is attributed not only to increased tree mortality (Berenguer *et al* 2014, 2018), but also to shifts in forest composition following fires—which favor smaller, species that grow rapidly following disturbances but also die at higher rates (Brando *et al* 2019, Esquivel-Muelbert *et al* 2019).

Climate models suggest that the southeast Amazon will experience prolonged droughts in the coming years (Duffy *et al* 2015, Marengo and Espinoza 2016), leading to a significant increase in the frequency and intensity of fires (Fonseca *et al* 2019, Uribe *et al* 2023). When forest fires occur under extreme drought events, they tend to be more intense due to higher fuel loads (dead components) and dryness (Brando *et al* 2014, 2019), reducing forest biodiversity (Barlow *et al* 2016) and thus contributing to a detrimental climate feedback loop (Cochrane and Laurance 2002). Given this scenario, we can expect significant changes in both aboveground carbon stocks and forest composition as fires become more widespread and severe.

Experimental studies are fundamental tools for understanding the complex dynamics of such fire disturbances. These experiments have permitted the manipulation of fire frequency, fuel loads, and fire intensity in controlled conditions, allowing direct observation of the short and long-term fire responses of species diversity, forest composition, and carbon stocks. They can thus be used to simulate potential future disturbance scenarios with frequent and intense fires, and to predict the capacity of Amazonian forests to resist and recover from such events (Brando *et al* 2016, Aragão *et al* 2018). The unique datasets generated in such studies are essential for developing effective management and conservation policies (Barlow *et al* 2018).

In this study, we used experimental forest fires to address two questions: (i) to what extent are tree diversity and species composition in tropical forests resilient to repeated fires of different frequencies? (ii) What are the impacts of these disturbances on aboveground carbon stocks, considering that fires cause the death of large trees that release large amounts of carbon, while fostering regeneration dominated by smaller, fast-growing species? Addressing these questions is crucial for understanding forest resilience, as well as predicting and mitigating the long-term impacts of fires on tropical forest ecosystems. Taking advantage of an ongoing fire experiment in Southeastern Amazonia, we tested the following hypotheses: (i) burned forests will show lower tree species diversity and changes in species composition compared to unburned plots, due to fire-induced tree mortality; (ii) forests experiencing more frequent and intense fires will show slower recovery of species diversity and composition; (iii) fire treatments with added fuel and higher fire frequency will experience greater reductions in tree diversity and carbon stocks, along with changes in species composition, owing to increased fire severity; (iv) burned forests will exhibit long-term declines in aboveground carbon stocks relative to unburned plots.

2. Materials and methods

2.1. Study area

This study was conducted at the Tanguro Research Station, located on the Tanguro farm in the municipality of Querência, Mato Grosso, Brazil (figure 1). The site is situated in the transitional zone between the Amazon and Cerrado biomes. The vegetation is predominantly composed of seasonally dry evergreen forests with a relatively low canopy (20 ± 1 m; mean ± SE) compared with more humid Amazon forests to the north and west (30 ± 1 m; (Balch *et al* 2008)). Currently, approximately 60% of Tanguro farm consists of primary forests, while the remaining 40% have undergone significant land-use changes. Forests were converted to pasture beginning in 1976, followed by soybean croplands in the early 2000s, with recent shifts to continuous cropping of soybean, corn, and cotton (Maracahipes-Santos *et al* 2020). The average annual air temperature varies between 24 °C and 26 °C. Annual rainfall ranges from 1700 to 2200 mm, with a marked dry season from May to September (Alvares *et al* 2013), when rainfall events ⩾10 mm are rare (figure S1).

2.2. Experimental design

In 2011, we established six experimental randomized blocks with four treatments, totaling 24 plots of 40 × 40 m. The experimental fires were conducted following similar procedures in August of 2013 and 2016. This ensured consistency in the design

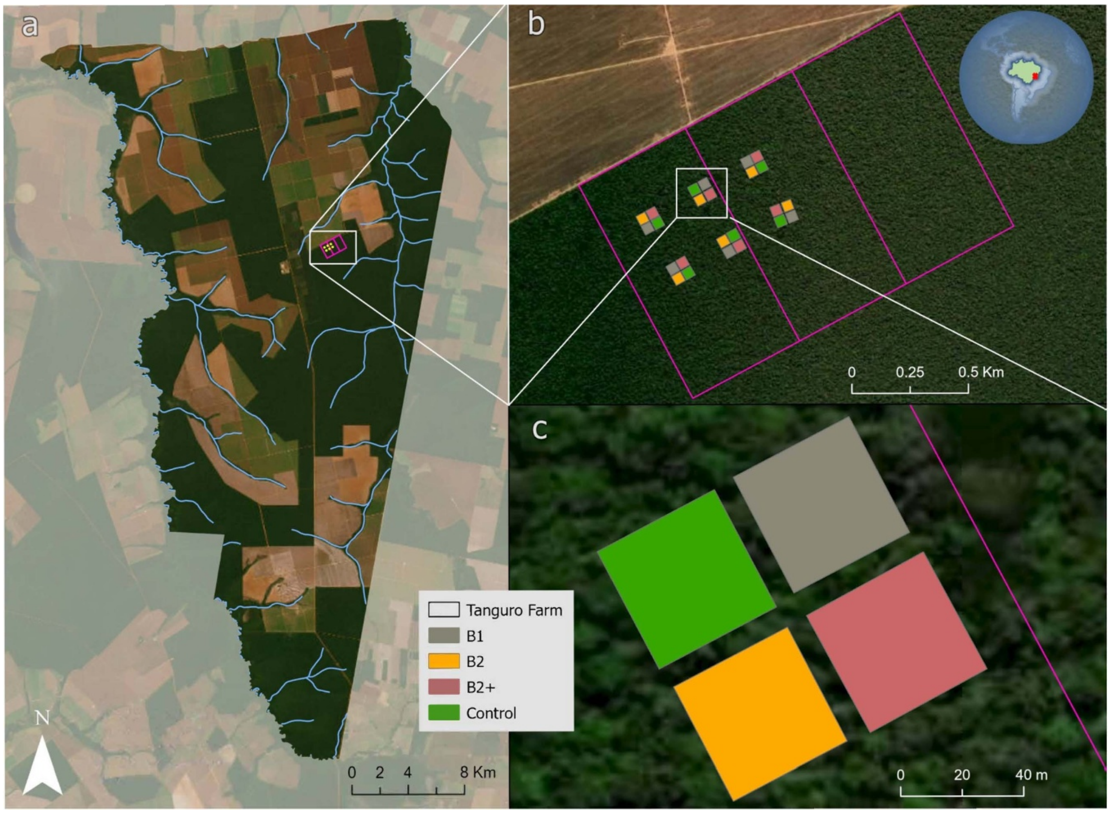

Figure 1. Location of the experimental fire plots in the Amazon–Cerrado transitional forest (Querência–MT, Brazil). (a) Location of the plots within the Tanguro farm; (b) diagram of the six experimental block and their location within the Amazon (top right); (c) spatial distribution of treatments within each randomized block, composed of four 40 × 40 m plots. B1 = plots burned experimentally only in 2016; B2 = plots burned experimentally in 2013 and 2016; B2+ = plots burned experimentally in 2013 and 2016 with added fuel; control = unburned plots. All maps were created with ArcGIS Pro v. 3.5 (ESRI, California, USA; Esri 2025).

and implementation of the experiment. This experimental area had no recent record of fire or selective logging activity, and is characterized by a low slope (<2%) (Brando *et al* 2016). Details on fire behavior can be found in table 1 of Brando *et al* (2016) and in the text of this manuscript (table 1). Briefly, the fire experiment consisted of four treatments: an unburned plot (control); plots that were burned for the first time in 2016 (B1); plots that were burned twice (2013 and 2016) with no added fuel (B2); and plots that were burned twice (2013 and 2016), following experimental fuel additions (B2+) approximately 3.2 Mg ha^{-1} collected from an adjacent forested area (see more details in Brando *et al* 2016). All prescribed fires were conducted between 12:30 PM and 1:30 PM using drip torches. In cases when the fireline was extinguished during this period, we reignited it up to 10 times.

2.3. Ecological measurements

In each plot, we conducted annual forest inventories during the dry season, measuring all living trees (i.e. trees, lianas, and palms) with diameter at breast height (DBH) ⩾5cm at 1.3 m height from 2012 to 2016, and subsequently in 2018, 2022 and 2024. In 2013 and 2016, the inventories were conducted within one month prior to the experimental fires, ensuring consistent environmental conditions across plots during data collection and providing nearly a full year of post-fire observation before the next scheduled burn. For each sampled tree, we measured the total height (m) and DBH, and recorded its position within the plot (*X* and *Y* coordinates).

We used forest inventory data to estimate aboveground biomass (AGB) and net primary productivity. Aboveground biomass was calculated using allometric equations developed for tropical forests (Chave *et al* 2014), incorporating tree DBH, height, and species-specific wood density, as implemented in the 'computeAGB' function from the R package BIOMASS (Réjou-Méchain *et al* 2017). We analyzed data from three census intervals (2012–2016, 2017–2018, 2019–2022 and 2023–2024), each of which included all surviving individuals as well as any new recruits (>9.9 cm in DBH). Using data from annual inventories, we tracked changes over time, normalizing each year's biomass by the initial year of 2012. We also compared the AGB of each year's experimental burn treatments with the control treatment.

Table 1. Fire intensity metrics in the Amazon–Cerrado transitional forest (Querência–MT, Brazil) across four experimental treatments: plots burned in 2016 [B1]; 2013 and 2016 [B2]; 2013 and 2016 with added fuel [B2+]; and unburned [control].

Acronyms	Fire intensity metrics (units)	B2+ (2013)	B2 (2013)	*P* value	B2+ (2016)	B2 (2016)	B1 (2016)	*P* value
R	Rate of fire spread (m min^{-1})	0.22 (0.17, 0.27)	0.23 (0.20, 0.26)	ns	0.24 (034, 0.17)	0.25 (0.34, 0.18)	0.25 (0.31, 0.18)	ns
FH	Flame height (cm)	34 (23, 48)	24 (20, 28)	**<0.05**	26 (44, 11)	25 (28, 20)	23 (33, 16)	ns
FL	Flame length (cm)	38 (27, 52)	29 (24, 33)	ns	30 (53, 15)	31 (36, 26)	29 (37, 22)	ns
FW	Flame width (cm)	17 (13, 22)	15 (11, 20)	ns	14 (18, 5)	14 (23, 10)	13 (18, 9)	ns
AB	Area burned (%)	88 (86, 90)	68 (62, 75)	**<0.001**	57 (67, 39) *	69 (82, 31)	75 (91, 62)	**<0.001**
FI	Fireline intensity (kW m^{-1})	76 (56, 100)	63 (52, 72)	ns	51 (76, 22)	60 (89, 34) *	27 (66, NA)	**0.045**

The 95% bootstrap confidence (lower, upper) interval is in parentheses.

ns = not significant; * = differed significantly; Bold = differed significantly.

In addition, we estimated aboveground net primary productivity (Clark *et al* 2001) based on changes in biomass between inventories (2012–2016, 2017–2018, 2019–2022 and 2023–2024 following Brando *et al* (2016) and Clark *et al* (2001).

In each plot, four 60 × 80 cm trays were used to collect litterfall samples every 15 d, which were then dried in a forced air circulation oven at 65 °C for 72 h. We also estimated leaf area index (LAI; $N = 5$ per plot) monthly using two LAI-2200C Plant Canopy Analyzers (Li-COR Biosciences Inc, Lincoln).

2.4. Statistical analysis

To compare tree species richness among the four treatments, we constructed rarefaction curves. We standardized the sampling effort by the number of individuals in the sampled area (Gotelli and Colwell 2001), using the *iNEXT* function from the '*iNEXT*' package in R version 4.4.1 (Chao *et al* 2014, Hsieh *et al* 2016, R Core Team 2024). We extracted the number of tree species estimated by the rarefaction curves for direct comparison of species richness between the control and experimental burn plots, after standardizing the sampling effort by individuals. We then compared species richness between plots using an analysis of variance (ANOVA). To compare changes in tree species composition between plots, we applied non-metric multidimensional scaling (NMDS) using the Bray–Curtis dissimilarity index (Legendre and Legendre 2012) and confirmed the significance of the changes using an ANOVA test. Ordinations were performed with both two ($k = 2$) and three ($k = 3$) dimensions. We evaluated the quality of fit using the stress value and Procrustes residuals. Although $k = 2$ provided interpretable configurations, stress values were consistently lower with $k = 3$ (range: 0.114–0.140 vs 0.166–0.197), indicating better model performance. Thus, we present ordination results based on $k = 3$ in the main text.

For a general characterization of the plots and blocks, we evaluated the time series of litter production and LAI. To assess treatment effects across time, we grouped the years into four fire phases based on the experimental timeline and fire history: (i) 'Before'(⩽2012), representing the pre-fire period; (ii) 'Post2013'(2013–2015), after the first burn; (iii)'Post2016' (2016–2018), following the second fire event; and (iv)'Post2018' (>2018), representing the longer-term post-fire recovery phase. We used a linear mixed-effects model (LMM) to evaluate the effects of fire phase. To minimize the effects of different sensor on LAI, we normalize all values within block by the control. Our linear statistical model included fixed effects for *FirePhase*, Treatment, and their interaction, and a random intercept for Block to account for spatial structure. We calculated the plant mortality rate for each treatment and used a generalized LMM to compare plant mortality rates across treatments and between years.

3. Results

Fire behavior varied across treatments in our experimental forest located in the Amazon–Cerrado transition zone. Plots with added fuel (B2+) consistently exhibited modest but measurable increases in fire intensity (table 1). In 2016, for instance, B2+ plots experienced significantly higher fireline intensity (51 kW m^{-1}) and a greater proportion of area burned (57%) compared to the single-burn (B1) and repeated-burn (B2) treatments ($P < 0.05$) (table 1). During the 2013 burns, B2+ plots also showed significantly taller flame heights and more extensive burned area than B2 (table 1). These results indicate that fuel additions intensified fire behavior, resulting in a more intense fire regime in B2+ plots. This increased intensity provides important context for interpreting the ecological responses presented in the following sections.

Overall, the forest plots subjected to experimental fires experienced changes in plant species diversity, composition, and biomass compared with unburned plots. In the pre-treatment period of 2012 and 2013, tree species diversity and composition were similar across the four treatments (control, B1, B2 and B2+; figures 2, S2 and 3). In the following years, the control and B1 treatments remained similar until 2016, when B1 was burned for the first time (figures 2, S2 and 3). In contrast, B2 and B2+ treatments exhibited gradual changes in tree species diversity, composition, and number of individuals from 2014 to 2024 (figures 2, S2 and 3), reflecting the prolonged impact of higher fire frequency and intensity. Similarly, treatment B1 showed gradual changes from 2018 to 2024 (figures 2, S2 and 3). NMDS ordinations provided consistent and robust representations of floristic dissimilarity across treatments and years (figure 3; table S1).

Aboveground forest biomass in the burned plots showed small reductions compared to the control plot (figure S3). The reduction in forest AGB was more evident when we normalized by the initial biomass in 2012 (figure 4(A)) or by the biomass in the control plot over the years (figure S4), but there was no ecological relevant difference (table S5).

Aboveground net primary productivity (ANPP) exhibited considerable interannual variability between 2012 and 2016, with values ranging from below 4 Mg ha^{-1} yr^{-1} (e.g. in 2013–2014) to peaks around 5 Mg ha^{-1} yr^{-1} in some treatments (notably B2+ in 2014–2015) (figure 4(B), table S6). Following the second fire event in 2016 (B2 and B2+), all treatments experienced a decline in ANPP, which became

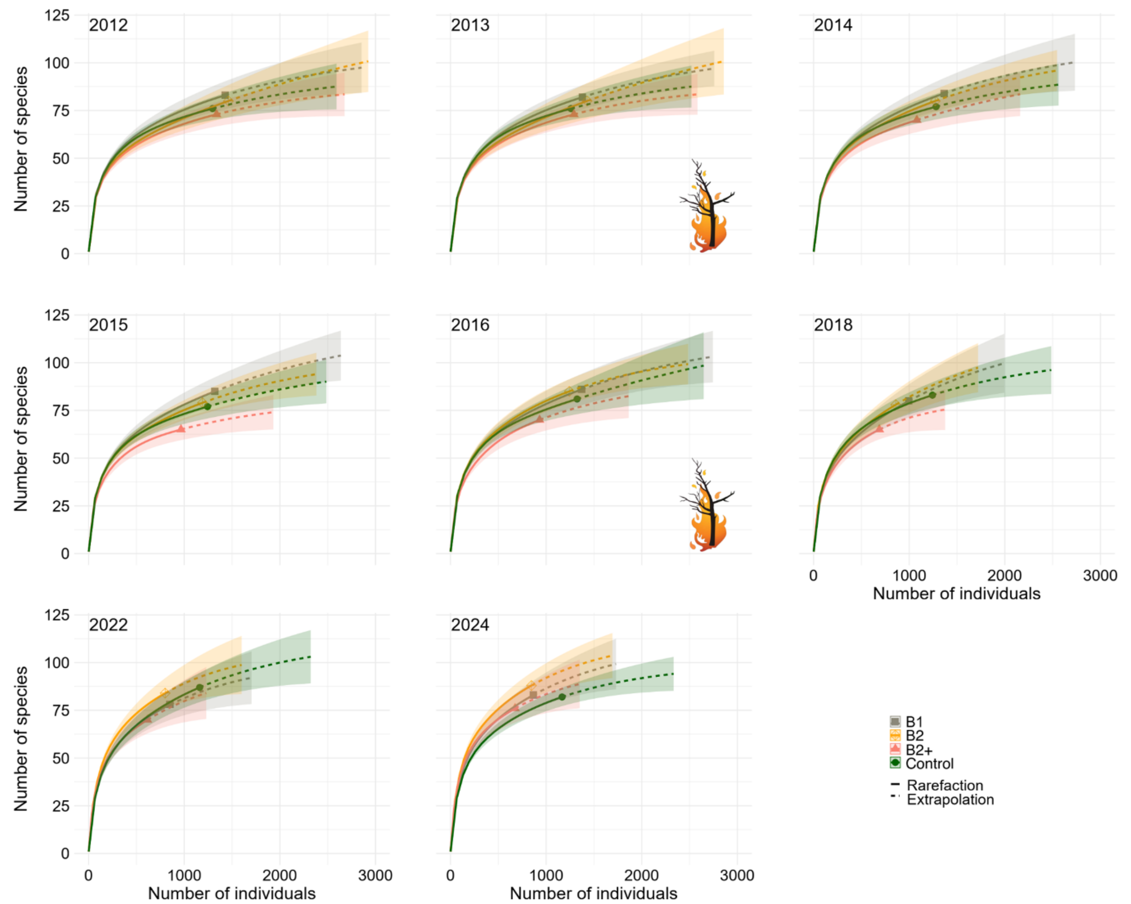

Figure 2. Diversity of woody species in an Amazon–Cerrado transitional forest (Querência–MT, Brazil) across four experimental treatments: plots burned in 2016 [B1]; 2013 and 2016 [B2]; 2013 and 2016 with added fuel [B2+]; and unburned [control]. The flaming tree graphics denote years with experimental burns.

more pronounced in the subsequent interval (2018–2022), particularly in the control plots, where values approached 3 Mg ha^{-1} yr^{-1} (figure 4(B)). Burned treatments also declined but to a lesser extent. During the final period (2022–2024), ANPP values stabilized at lower levels across all treatments, remaining below those observed in the early years of the study (e.g. 2012–2013), but with less inter-treatment variation (figure 4(B), table S6).

In general, burned plots exhibited LAI values comparable to control plots prior to and shortly after the fire events, suggesting initial structural similarity among treatment plots (figure 5(A)). Furthermore, positive shifts in ΔLAI during the Post2016 and Post2018 phases suggest a substantial recovery of canopy structure over time, particularly in plots with lower fire intensity (B1) (table S2; figure S5). In contrast, plots exposed to repeated or more intense burns (B2, and B2+) took longer to recover LAI, especially in certain blocks (e.g. EF2; table S2; figure S5).

In the early years (before and after 2013) litterfall control plots than in the burned treatments, possibly reflecting increased leaf turnover in recovering forests (figure 5(B); table S3). Over time, differences among treatments narrowed, but there was a high variability across blocks (table S3; figure S6). Notably, plots under higher fire intensity (B2+) often maintained elevated litterfall levels relative to controls, suggesting sustained disturbance-induced stress or compensatory growth dynamics (table S3; figure S6). The average annual litter production did not differ significantly among treatments (figures S7 and S8).

The mortality rate significantly increased after fire events in treatments B2 and B2+, especially after the 2014 inventory, and after the first burn of B1 in 2018 (figure 6; table S4). The B2+ treatment, with two burns and the addition of fuel, showed the highest mortality rates and greatest variation (figure 6; table S4). In general, trees with DBH $<$ 20 cm died at much higher rates than the large ones. Mortality rates in the control plot remained relatively low and constant over the years (figure 6; table S4).

4. Discussion

In this study, we experimentally assessed the impacts of different frequencies and intensities of wildfires in southeast Amazonia. Fires significantly affected plant

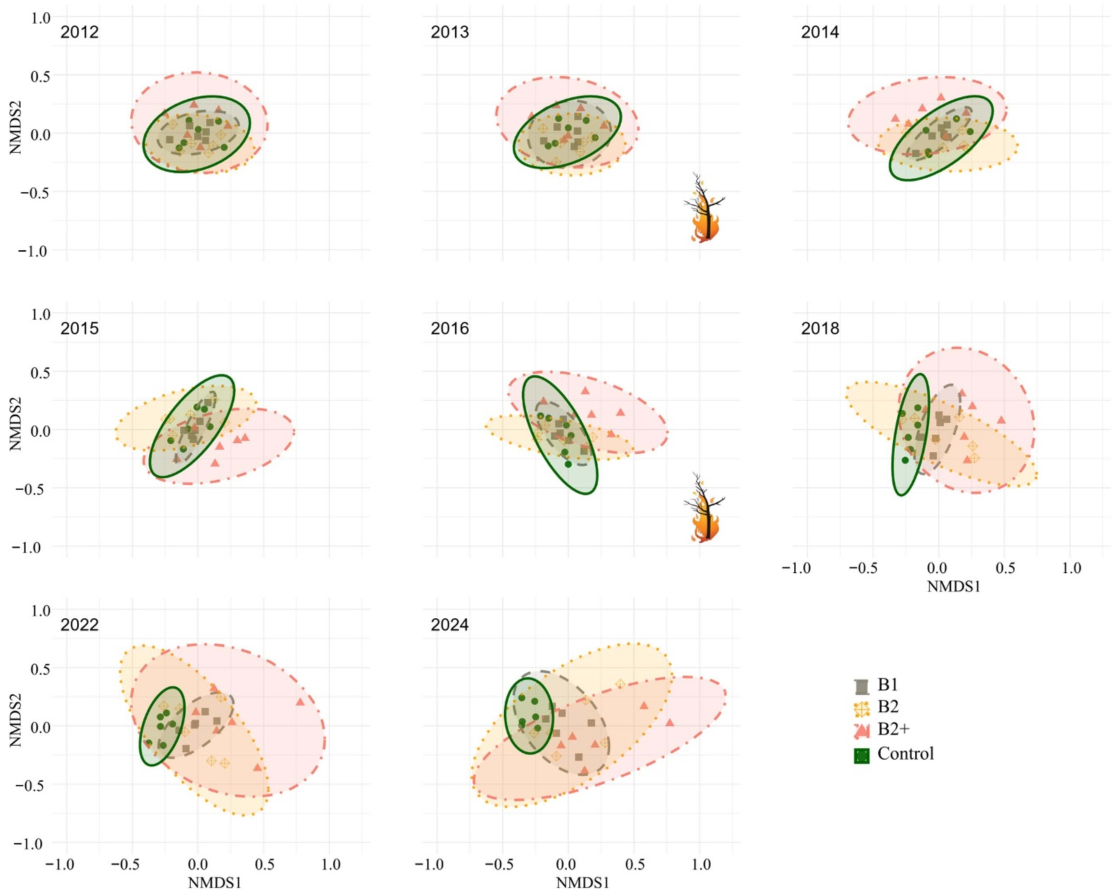

Figure 3. Composition of woody species in the Amazon–Cerrado transitional forest (Querência–MT, Brazil) across four experimental treatments: plots burned in 2016 [B1]; 2013 and 2016 [B2]; 2013 and 2016 with added fuel [B2+]; and unburned [control]. The flaming tree graphics indicate years with experimental burns. NMDS = Non-metric multidimensional scaling ($k = 3$; stress values ranging from 0.114 to 0.140).

species diversity, composition, and forest structure, particularly in plots with added fuel that experienced high-intensity fires. This suggests that wildfires could exert long-term influences on the structure, composition, and functioning of forests, even when they have some degree of resilience, as shown in our experiment. While the projected intensification of Amazon fire regime climate change may further jeopardize forest resilience and ecosystem services, our study shows that the transitional forests between Amazonia and Cerrado have the capacity to cope with amount of disturbance by repeated fires.

Results from our long-term experimental fire studies in the Amazon reveal that low-intensity fires can lead to elevated mortality of small trees and modest changes in species richness and composition, but do not necessarily result in catastrophic forest loss. These findings highlight a degree of forest resilienceto isolated, low-intensity fire events, though the observed impacts are ecologically relevant. However, projections that up to 16% of forests in the southeastern Amazon—particularly in its drier zones—could burn in the coming decades, it is uncertain whether this resilience persist in the face of more frequent and intense fire regimes (Berenguer *et al* 2014, Brando *et al* 2020). A key insight from our study is the role of increased fuel loads in promoting fire continuity and expanding the total burned area, which in turn amplifies tree mortality. As droughts and heatwaves become more frequent, these interactions may intensify (Flores *et al* 2024). Edge effects could further compound fire severity. Previous studies in the region have shown that fires along forest edges, where conditions are drier and hotter, can kill up to 90% of trees after repeated burns—significantly more than in forest interiors similar to the experimental forests in our study (Brando *et al* 2014).

This finding is consistent with previous studies showing that tropical forests in southeast Amazonia tend to be resistant to low-frequency, and low-intensity fires (Brando *et al* 2014, 2016, 2019, Pereira *et al* 2024). However, a few years after the fire event, the structure and composition of the forest in this treatment had changed significantly (figure 3; table S1). The long-term impacts of the experimental fires were even more pronounced in the experiments with increased fire frequency (B2) and fuel additions (B2+), mainly in terms of reduced number of surviving individuals and increased tree mortality. Higher-intensity fires (B2+) resulted in tree

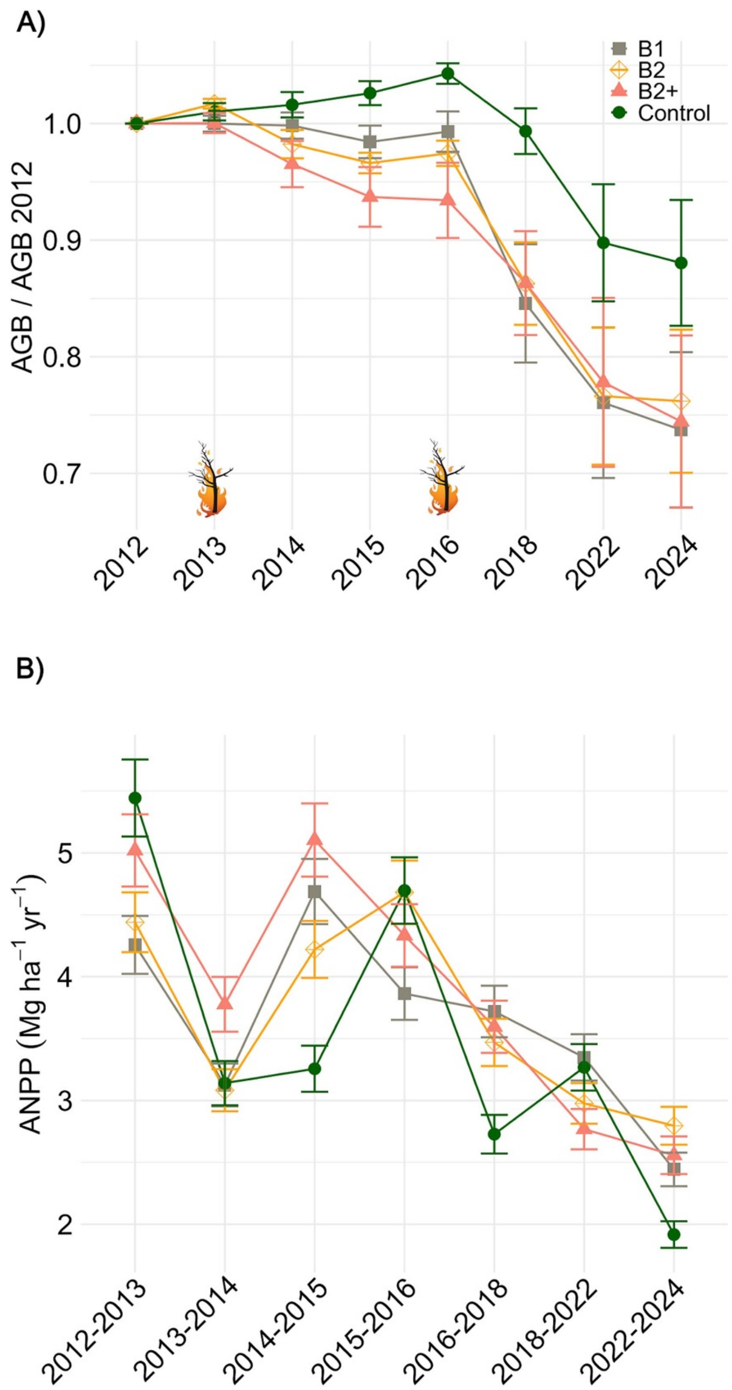

Figure 4. Total aboveground biomass (AGB) divided by initial biomass in 2012 (A), and aboveground net primary productivity (ANPP; B) in the four experimental treatments in an Amazon–Cerrado transitional forest (Querência–MT, Brazil). Treatments consisted of: plots burned in 2016 [B1]; 2013 and 2016 [B2]; 2013 and 2016 with added fuel [B2+]; and unburned [control]. Flaming tree graphics indicate the timing of experimental burns. Bars indicate 95% confidence intervals.

mortality rates four to five times higher in 2014 than in the control and B1 plots, demonstrating that these fires have a greater impact on tropical forests. Given that droughts tend to increase fuel loads, our results support previous studies showing that wildfires are more intense and severe during droughts due to both increased air dryness and higher fuel loads. Our findings underscore that tree mortality continues to impact degraded forests over the long term, since trees in these conditions are vulnerable to mortality from secondary disturbances (e.g. severe droughts and blowdowns; Phillips *et al* 2009, Negrón-Juárez *et al* 2018, Silvério *et al* 2019).

Despite important changes in forest dynamics, our study also showed that fires had limited impacts on forest structure at the stand level. Despite a 14% (B1), 11% (B2), and 13% (B2+) reduction in aboveground carbon stocks, the ANPP and LAI in the

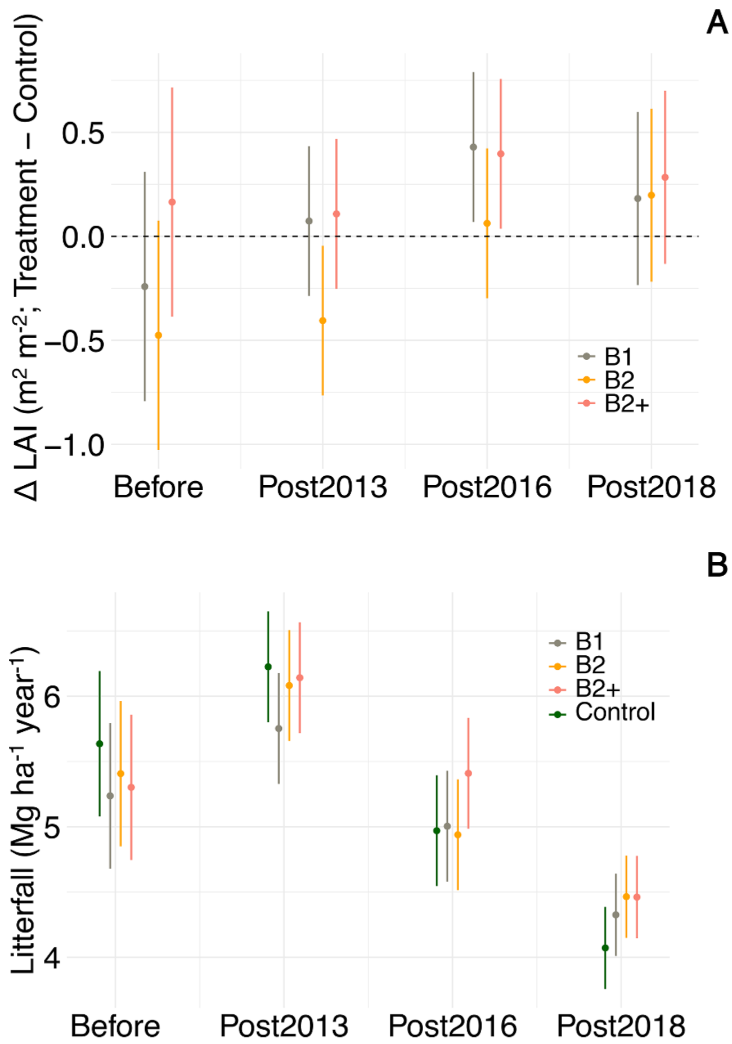

Figure 5. (A) Leaf area index (LAI), and (B) average annual litterfall, across four experimental treatments in an Amazon–Cerrado transitional forest (Querência–MT, Brazil). Treatments consisted of: plots burned in 2016 [B1]; 2013 and 2016 [B2]; 2013 and 2016 with added fuel [B2+]; and unburned [control]. Flaming trees denote the timing of experimental burns. Bars indicate 95% confidence intervals.

burned treatments remained similar to the base line year of 2012, suggesting a partial recovery of ecosystem services (e.g. carbon flux) eight years after the last fire. Notably, ANPP values declined across all treatments (including the control) after repeated burns, but recovered in the final study period. These results are consistent with an other study showing that carbon fluxes and evapotranspiration in burned forests can recover completely within a decade after the last fire event (Brando *et al* 2019). These findings highlight the resilience of burned forests partially recovered key ecosystem functions such as carbon storage and species diversity within a decade following fire disturbances.

Patterns in LAI and litterfall reveal substantial spatial and temporal heterogeneity in ecosystem responses to fire, shaped by both fire history and local environmental conditions. LAI declined after each burn but rapidly recovered, highlighting the capacity to recover photosynthetic capacity and forest structure. However, in previous studies, repeated fires drove much steeper reductions in canopy cover (Brando *et al* 2014). One explanation for these contrasting results is the lack of forest edges in our experimental setting. Overall, forest edges adjacent to agricultural fields are more degraded and susceptible to fires (Cochrane and Laurance 2002, Brando *et al* 2014, 2019, Silvério *et al* 2019). As such, the absence of forest edges in this study potentially contributed to the rapid recovery of some ecosystem processes (e.g. canopy cover) via tree recruitment. However, we cannot discard the possibility that the reductions in biomass and LAI observed in the control plot could be due to indirect effects from burning adjacent plots or regional drying. Our findings reinforce the notion that forests are resilient to low-frequency fires, but underscore the prolonged recovery (Brando *et al* 2019) period required after intense fires and secondary disturbances.

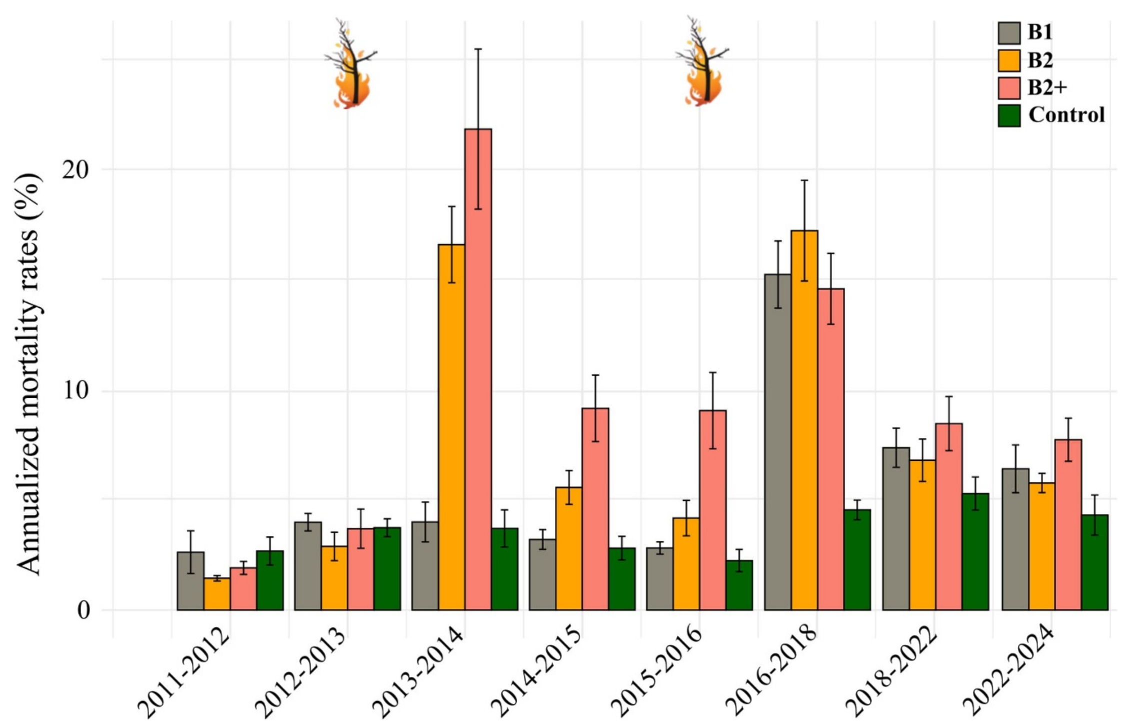

Figure 6. Annualized mortality rates across four experimental treatments in an Amazon–Cerrado transitional forest (Querência–MT, Brazil). Treatments consisted of: plots burned in 2016 [B1]; 2013 and 2016 [B2]; 2013 and 2016 with added fuel [B2+]; and unburned [control]. Flaming trees denote the timing of experimental burns. Bars indicate 95% confidence intervals.

Our results clearly indicated that the addition of fuel from dead components increased tree mortality. Climate change scenarios suggest that the forests of the southern Amazon are becoming warmer and that fuel loads will increase (Duffy *et al* 2015, Marengo and Espinoza 2016). This is largely due to changes in species composition, with evergreen species being replaced by deciduous species (Saatchi *et al* 2013) that increase leaf litter. We intentionally manipulated only one of these factors—the intensity of fires through fuel addition—and observed pronounced increases in tree mortality. Other studies have shown that severe droughts can also cause abrupt tree mortality, even without additional fuel (Brando *et al* 2014). Considering that forests have already experienced longer dry seasons, higher temperatures, lower humidity, and greater fuel availability, as predicted by climate change scenarios (Marengo *et al* 2018, Commar *et al* 2023), wildfires occurring under these conditions could have drastic impacts on forests (Brando *et al* 2014), with long-term implications for tropical carbon stocks and forest resilience.

An important question is whether Amazonian forests impacted by fire can recover their structure, composition, and function. Our study shows that when some drivers of forest degradation (e.g. edge effects, grass invasion) are removed, these ecosystems are sufficiently resilient to low-intensity fires. Although forest resilience in our study can be considered high, recovery tends to start through the recruitment of fast-growing pioneer species with relatively thin bark (Poorter *et al* 2014) and low-density woods (Pinho *et al* 2024), which can make these trees vulnerable to future fire-drought events. All scenarios that increase the vulnerability of these forests, including climate changes and shifts in composition, pose significant challenges to the resilience and health of tropical forests (Trumbore *et al* 2015, Marengo *et al* 2018, Feng *et al* 2021). Maintaining the key ecosystem services provided by these forests demands urgent conservation measures and mitigation of fire impacts to protect forest health in the face of climate change.

5. Conclusions

Our results highlight that repeated and high-intensity fires can alter tree species composition in tropical forests. At the same time, our experimental sites showed signs of rapid recovery in forest structure and ecosystem function. However, climate change is likely to increase the frequency and duration of severe droughts in tropical forests, which can amplify fuel loads and, potentially, increase the intensity and severity of future fires. Although the study forests were resilient in the absence of compounding disturbances and seed dispersal sources, this resilience may be compromised under more extreme fire regimes and intensity. Rapid adoption of fire protection and management strategies is crucial to prevent the loss of key ecosystem services in tropical forests.

Data availability statement

All data that support the findings of this study are included within the article (and any supplementary files).

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Author contributions

The questions and research were developed by LM-S, LM, DVS, MNM, ACS, LNP, NP, BS, AACA, and PMB. Data collection was conducted by LM-S, DVS, ACS, NP, and LM. The analyses and the first draft of the manuscript were conducted by LM-S, LM, DVS, ACS, NP, BS, and PMB. All authors contributed substantially with reviews and approved the final manuscript.

Ethics statement

All authors have actively contributed to the manuscript, reviewed its content, and provided their agreement. Additionally, we have no conflicts of interest or financial disclosures to report.

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