

Wildfire impacts on vegetation structure in the Brazilian Atlantic Forest

WILDFIRE IMPACTS ON VEGETATION STRUCTURE IN THE BRAZILIAN ATLANTIC FOREST 8 YEARS AFTER A FIRE

Graziele Silva Cruz ^{1*} , Leonardo Lima Alves ¹ , Gabriela Alves Ferreira ^{2, 4} , Tainá

Jardim Antunes³  & Pavel Dodonov ¹ 

1 Federal University of Bahia, Institute of Biology, Spatial Ecology Laboratory, Postgraduate Program in Ecology: Theory, Application, and Values, Rua Barão de Jeremoabo 668/Ondina, 40170-115, Salvador, Bahia, Brazil.

2 State University of Santa Cruz, Tropical Herpetology Laboratory, Postgraduate Program in Ecology and Biodiversity Conservation, Jorge Amado, Km 16/Salobrinho, 45662-900, Ilhéus, Bahia, Brazil

3 Federal University of Southern Bahia, Applied Forest Ecology and Silviculture Laboratory, Postgraduate Program in ecology and conservation of biodiversity, Rodovia Ilhéus/Itabuna, Km 22, 45604-811, Ilhéus, Bahia, Brazil

4 State University of Santa Cruz, Applied Ecology and Conservation Laboratory, Postgraduate Program in Ecology and Biodiversity Conservation, Jorge Amado, Km 16/Salobrinho, 45662-900, Ilhéus, Bahia, Brazil

E-mails: grazielesccruz@gmail.com (*corresponding author); leonardolima0398@gmail.com; gabriela-alves77@hotmail.com; tjantunes.ppgecb@uesc.br; pdodonov@gmail.com

Abstract: Wildfires are a severe threat to biodiversity, especially in fire-sensitive ecosystems such as tropical rainforests. We assessed differences in vegetation structure between burned and unburned Brazilian Atlantic Forest fragments eight years after a wildfire. We expected higher tree mortality, a greater abundance of ferns, and reduced canopy cover in burned areas, reflecting limited forest recovery. Although the number and basal area of snags (standing dead trees) did not differ between areas, burned forests had fewer living trees, more snags per plot, and a greater number of fallen trees. Living trees in unburned areas exhibited larger basal areas, whereas snags were larger in burned areas. Canopy cover was lower and more variable in burned sites. In the

understory, Neotropical bracken fern dominated burned forests, while herbaceous plants and palms were more frequent in unburned areas. The results indicate that fire impacts persist nearly a decade after disturbance, as evidenced by the higher abundance and basal area of standing and fallen dead individuals, confirming fire as a key agent of structural alteration in tropical forests. Additionally, the dominance of superdominant species and the low canopy cover may compromise natural regeneration. However, the remaining snags, as well as the living trees, in the burned forest may act as perches, facilitating seed rain. Concluding, our results indicate that natural regeneration is not sufficient, and active restoration strategies should be employed for the recovery of burned Neotropical rainforests.

Keywords: Forest degradation; Forest fire; Tropical forest; Overabundant species; Natural regeneration.

INTRODUCTION

Tropical forests are home to nearly half of all terrestrial biodiversity worldwide (Pimm and Raven, 2000) and contribute to one-third of land-surface productivity and evapotranspiration (Malhi, 2012). Despite their biological importance for the planet, these forests are in decline worldwide due to threats such as deforestation, climate change, species invasion, spread of pathogens, and wildfires (Malhi *et al.* 2014). This scenario is becoming concerning because most of these impacts occur together or in sequence. For instance, the increase in wildfires can intensify carbon emissions and contribute to climate change. Similarly, wildfires may have synergistic interactions with habitat loss and fragmentation (Driscoll *et al.* 2021).

Tropical rainforests, such as the Atlantic Forest in Brazil, have no indication of evolutionary history influenced by fire, and therefore, their species may not have adaptations that favor their resistance to fire or their recovery post-fire (Pivello *et al.* 2021). Fire causes tree

mortality, opening the canopy, reducing humidity, and favoring pioneer trees that produce large amounts of litter to occupy the gap, which increases the forest's susceptibility to subsequent fire events (Sansevero *et al.* 2020). Furthermore, extreme droughts increase the probability of forest fires. The occurrence of fire in these forests drives environmental changes that reduce the suitable habitat and resources for animal species (Driscoll, 2021). Additionally, changes in forest structure and composition affect fruit production for large vertebrates (Barlow and Peres, 2006) and alter patterns of bird and dung beetle diversity (de Andrade *et al.* 2014; Hidasi-Neto *et al.* 2012).

Although tropical forest communities are resilient to various disturbances, fires represent a particularly severe threat as they impact the abundance, diversity, structure, and regeneration patterns of forest species (Buragohain *et al.* 2023; Cury *et al.* 2020; Menezes *et al.* 2019). These fires can damage both standing vegetation and soil layers containing seed banks (Balch *et al.* 2013; Scheper *et al.* 2021), as well as seed rain and plant litter production (Christo *et al.* 2021; Rocha *et al.* 2022, 2024). However, when soil seed banks remain intact after a fire, recovery can be successful, as many understory species in tropical forests regenerate from seeds stored in the soil (Baker *et al.* 2013; Scheper *et al.* 2021). Standing dead trees may also aid regeneration, as they may act as perching sites for seed-dispersing birds (Gama *et al.* 2025). For instance, a study comparing burnt and unburnt Atlantic Forest fragments in southeastern Brazil found similar vegetation structure and functional traits between them nine years after the fire, although species composition differed (Carvalho *et al.* 2022). Conversely, another study detected relevant differences between a burnt and an unburnt Atlantic Forest site 16 years after a fire (Christo *et al.* 2021). Thus, assessing fire impacts at different times after the disturbance is important for a better understanding of the forest's natural regeneration capacity.

The Atlantic Forest, recognized as one of the world's five major biodiversity hotspots, harbors rich diversity, despite only 23% of its original extent remaining (Tabarelli *et al.* 2005; Vancine *et al.* 2024). Most of the remaining fragments consist of small, isolated areas

predominantly composed of secondary forests in early or intermediate stages of succession (Metzger *et al.* 2009; Ribeiro *et al.* 2009; Vancine *et al.* 2024). The Atlantic Forest of southern Bahia contains some of the largest remnants in northeastern Brazil and exhibits high levels of plant endemism (Benchimol *et al.* 2017; Rocha-Santos *et al.* 2017). These forests have a long history of fragmentation, leading to structural and compositional changes that are linearly related to the amount of forest cover (Faria *et al.* 2025; Rocha-Santos *et al.* 2017). Consequently, forest remnants are particularly susceptible to fire, a disturbance that not only reduces species diversity but also alters forest composition and structure, thereby compromising ecological processes and forest regeneration (Sansevero *et al.* 2017).

Analyzing vegetation structure in regenerating burned areas is essential for understanding the mechanisms of ecological recovery following fire disturbances (Buma & Wessman, 2012). Between January and June 2024, more than 3,000 hectares of the Atlantic Forest were impacted by wildfires (MapBiomass, 2024), highlighting the importance of studies that assess the effects of such disturbances on the biome. Fire favors species adapted to disturbance conditions while negatively affecting more sensitive species, disrupting successional dynamics and altering the composition of plant communities (Fisher *et al.* 2009). Such disturbances may also modify the size structure of tree populations (Eichenwald *et al.* 2025). For example, ferns such as *Pteridium* sp. often expand after fire events due to their vigorous rhizomatous growth, the survival of buried spores after burning events, and rapid occupation of newly opened areas (Trejo *et al.* 2010; Paz *et al.* 2022). Similarly, bamboos may increase in burned forests because of their rapid clonal growth and resprouting ability, both potentially affecting successional trajectories and tree regeneration (Silva *et al.* 2021).

Investigating the structural regeneration of these areas enables the identification of successional stages, the evaluation of biodiversity changes, and the understanding of ecosystem resilience, which are critical for informing restoration and conservation strategies (Alves &

Metzger, 2006; Johnstone *et al.* 2016; Sansevero *et al.* 2017). For example, the simultaneous presence of live and dead trees—a structural variable—increases seed deposition in burned areas of the Atlantic Forest, thereby favoring forest regeneration when associated with suitable environmental conditions (Gama *et al.* 2025).

In this study, we tested for differences in the vegetation structure between burned and unburned forest in the Brazilian Atlantic Forest, a hotspot of biodiversity, eight to nine years after a wildfire. Considering the increasing frequency of wildfires and the need to better understand post-fire regeneration dynamics in tropical forests, our study aimed to assess natural regeneration in Atlantic Forest fragments nine years after a wildfire to assess whether fire impacts can still be observed after this period.

Fires are known to affect the size distribution of woody plants (Silva *et al.* 2009), and we expected the size distribution to differ between burnt and unburnt forests. We also expected to observe more dead trees, snags, and logs and fewer living trees in the burnt forest because of the high tree mortality caused by the fire (Menezes *et al.* 2019). In addition, we expected the living trees in the burnt forest to be on average smaller, as they are likely to be regenerating plants that were recruited after the fire, but the snags to be larger, due to high mortality of trees of different sizes in the fire, than in the unburnt forest. Due to tree mortality, we also expected that the burned forests would have lower canopy cover values. Regarding the understory, we expected to observe a greater occurrence of the superdominant bracken fern *Pteridium esculentum* (G. Forst.) Cockayne and bamboos in burned forests, as has been previously observed (Menezes *et al.* 2019). These differences would indicate that the time span between the wildfires and the study was not sufficient for meaningful vegetation recovery.

METHODS

Study site

The study was conducted at the Refúgio de Vida Silvestre de Una (Revis de Una), a strictly protected conservation area within the National System of Conservation Units (Brazil, 2024). It encompasses 23,404 hectares and is located in the municipality of Una in southern Bahia, Brazil (15.08–15.16° S, 39.00–39.04° W) (Figure 1a). The Revis de Una includes a coastal zone with *restinga* (coastal scrub), mangroves, and Atlantic Forest fragments (Brazil, 2024). The region features a hot and humid climate with no defined dry season, annual rainfall ranging from 1,200 to 1,800 mm, and an average temperature of 24–25 °C (Menezes *et al.* 2019). The vegetation is classified as Tropical Humid Forest, characterized by predominantly yellow latosol soils (Rocha *et al.* 2024).

Data collection was conducted from September 28 to October 3, 2024, at three burned forest fragments undergoing early regeneration (areas of 28, 63, and 260 ha) and three adjacent unburned forest fragments (Figure 1b-f). The fires, triggered by agricultural activities and a severe drought during the 2015/2016 El Niño event, occurred from December 2015 to February 2016 (Dodonov *et al.* 2019) and lasted approximately one month in each fragment, significantly impacting soil and vegetation (Menezes *et al.* 2019; Rocha *et al.* 2024). Previous studies reported that these fires drastically altered vegetation structure, resulting in high tree mortality and promoting the dominance of the native superdominant fern *P. esculentum* (Menezes *et al.* 2019). Furthermore, a reduction in seed rain (Rocha *et al.* 2022) and litter production (Rocha *et al.* 2024), along with microclimatic changes (Dodonov *et al.* 2019), was observed. The presence of remnant trees in the burned area contributed to higher seed deposition beneath these trees (Gama *et al.* 2025).

Sampling design and data collection

We had between 10 and 13 randomly defined sampling points in each study site, with at least 30 m between adjacent points (Figure 1b-d). Vegetation data were collected within circular plots with a radius of 5 meters, centered at each sampling point. The circumference at breast

height (CBH, measured at 1.30 m above the ground) was recorded for all tree individuals with a circumference at breast height (CBH) ≥ 15 cm, including both living trees and standing dead trees (snags). When a tree had more than one stem, we measured all of its stems that had a CBH ≥ 15 cm. We also measured the diameter of the fallen dead trees (logs) approximately 1.30 m from their base or from the plot's limits. Total basal area and mean basal area were calculated as complementary structural metrics, representing the amount of woody biomass and the average size of individuals, respectively, allowing comparisons of vegetation structure between study sites. Canopy cover was estimated using a spherical canopy densiometer, with four perpendicular measurements at the center of plot. Additionally, the understory was examined by recording the presence or absence of bamboos, graminoids (grasses and sedges), *Pteridium esculentum*, other herbaceous plants, palms, lianas, and regenerating vegetation (woody individuals with a circumference at breast height < 3 cm), following the categories used by Menezes et al. (2019). This approach was adopted to ensure greater standardization among plots within a rapid assessment protocol, given that cover estimates at the sampling scale could be more prone to imprecision. In the case of *P. esculentum*, its occurrence is typically associated with the formation of dense patches (Menezes *et al.* 2019), a pattern also observed in the field, making this metric suitable for representing its dominance.

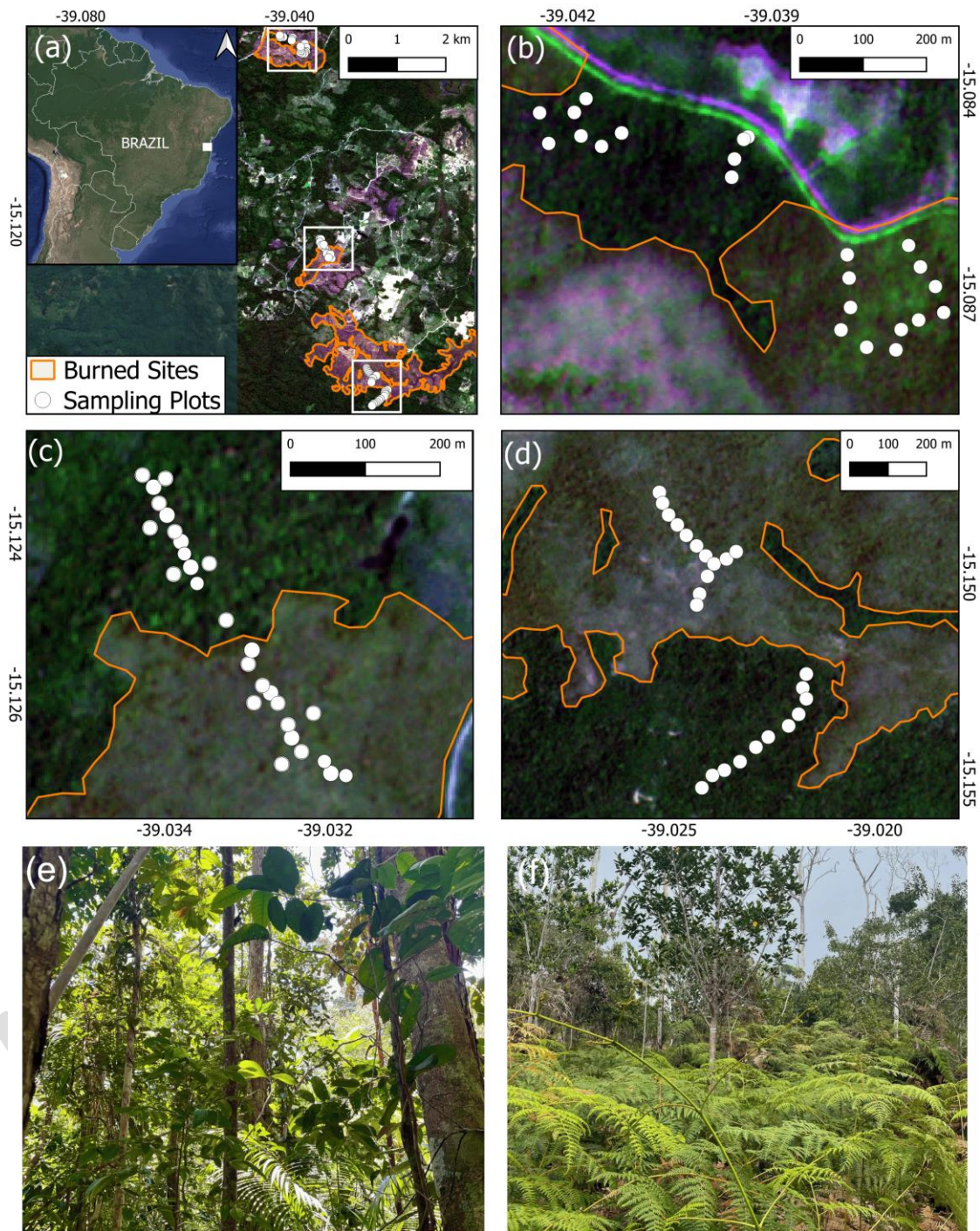


Figure 1. Location of the study area in the Refúgio de Vida Silvestre de Una, state of Bahia, Brazil. (a) Map showing the distribution of sampling plots (white circles) within burned (outlined in orange) and unburned forest. Forest fragments of 28 ha (b), 63 ha (c), and 260 ha (d). Representative field photographs of (e) unburned and (f) burned forests.

Statistical analysis

We used three groups of variables between the burned and unburned forest sites: arboreous vegetation structure, canopy cover, and understory structure. For canopy cover, we used the mean of the four measurements taken at each point. For the arboreous vegetation structure, we used the number of individuals, total basal area, and mean basal area per stem, separately for living trees, snags, and logs. The basal area was calculated as $(\text{Circumference}^2)/(4\pi)$, assuming that the trees were circular in cross-section. For multi-stemmed trees, we first calculated the basal area of each stem and summed these basal areas to obtain a single value for the tree. For understory structure, we used the presence or absence of each element (bracken fern; bamboos; graminoids; other herbaceous plants; palms; lianas; regenerating individuals) in each plot.

We first used a Kolmogorov-Smirnov test (Zar, 2010) to assess whether the circumference distribution of living trees, snags, and logs varied between the burned and unburned sites. We then used permutation tests (Manly, 2007) to test whether there were differences between the burned and unburned sites in the mean values and standard deviations (except for understory structure) of the response variables. For this, we first calculated the mean and standard deviation for each variable in both burned and not burned forests. We then computed the differences between the means and standard deviations of these groups. To test statistical significance, we randomized the response variable values, simulating the null hypothesis that the observed values could be randomly assigned to burned and unburned forests. This randomization was performed separately for each study fragment, meaning that the plots were randomized only between adjacent burned and unburned sites to account for the non-independence between them. Using randomized data, we recalculated the differences in means and standard deviations 9,999 times to generate a null distribution. Statistical significance was determined using a two-tailed test, where a difference was considered significant if the proportion of randomized differences was equal to or greater than the observed difference, using absolute values.

All analyses were performed in the R environment (R Core Team, 2022) using the following packages: *tidyverse* (Wickham *et al.* 2019), *ggplot2*, *ggthemes*, *scales*, *ggimage*, *ggpubr*, *gtable*, and *cowplot*. The functions used for the restricted randomization tests were developed by the authors and are available at <https://github.com/pdodonov/>.

RESULTS

Arboreous vegetation structure and canopy cover

We measured a total of 910 trees distributed across three burned and unburned forests. Of these, 731 were alive and 179 were dead, with 62 snags and 117 logs. The number of stems per tree varied from 1 to 5, but 97% of the living trees and all the snags and logs had a single stem. As expected, the fire resulted in differences in vegetation structure. There were more stems in the lower size classes, especially for the living trees, and the CBH distributions differed between burned and unburned forests for living trees and for snags but not for logs (Figure 2).

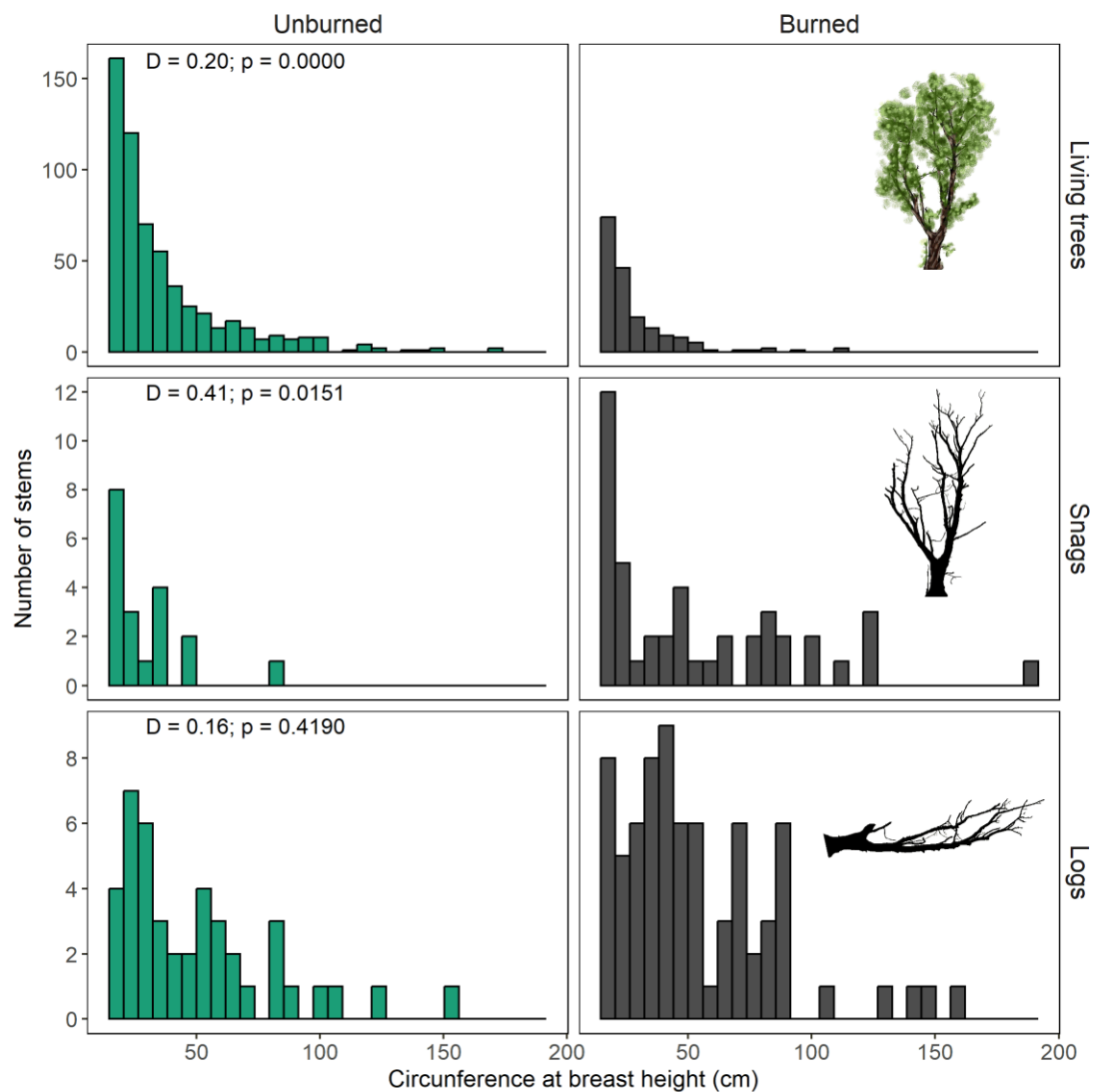


Figure 2. Distribution of stem circumference at breast height (cm) in burned forests and unburned forests for living trees, snags (standing dead trees), and logs (fallen dead trees), together with the Kolmogorov-Smirnov test statistics and p-values comparing the distributions between burned and unburned forest for each variable in the Refúgio de Vida Silvestre de Una, state of Bahia, Brazil.

The number of logs did not differ between the burned and unburned sites (Figure 3a). However, the burned sites had on average more snags and fewer living trees per plot than the unburned sites (Figure 3b-c). Specifically, the number of snags per plot in the burned forests was approximately twice that of the unburned ones (1.3 and 0.6 snags per plot, respectively), whereas

the number of living trees was approximately four times smaller (4.9 and 18.1 trees per plot, for the burned and unburned sites, respectively).

Similarly, the total basal area of logs did not differ between the burned and unburned sites (Figure 3d). The total basal area of snags per plot was greater in the burned sites than in the unburned sites (burned mean = 445 cm²/plot; unburned mean = 55 cm²/plot), whereas that of living trees was smaller (burned mean = 424 cm²/plot; unburned mean = 2960 cm²/plot) (Figure 3e–f). In contrast, the mean basal area of snags was greater in the burned sites (burned mean = 195 cm²/stem; unburned mean = 30 cm²/stem), whereas that of living trees was greater in the unburned sites (burned mean = 75 cm²/stem; unburned mean = 168 cm²/stem; $p < 0.001$). These results show that snags were, on average, larger in the burned forests, whereas living trees were larger in the unburned forests (Figure 3g–i).

In addition, the number of living trees per plot and their total basal areas were more variable in the unburned than in the burned sites, as shown by the significant difference in the standard deviations (Figure 3c). Conversely, the basal area of snags – both total basal area per plot and mean basal area per stem – was more variable in the burned sites (Figure 3e, h). There were no differences in the standard deviation between burned and unburned sites for any variable related to the logs (Figure 3a, d, g).

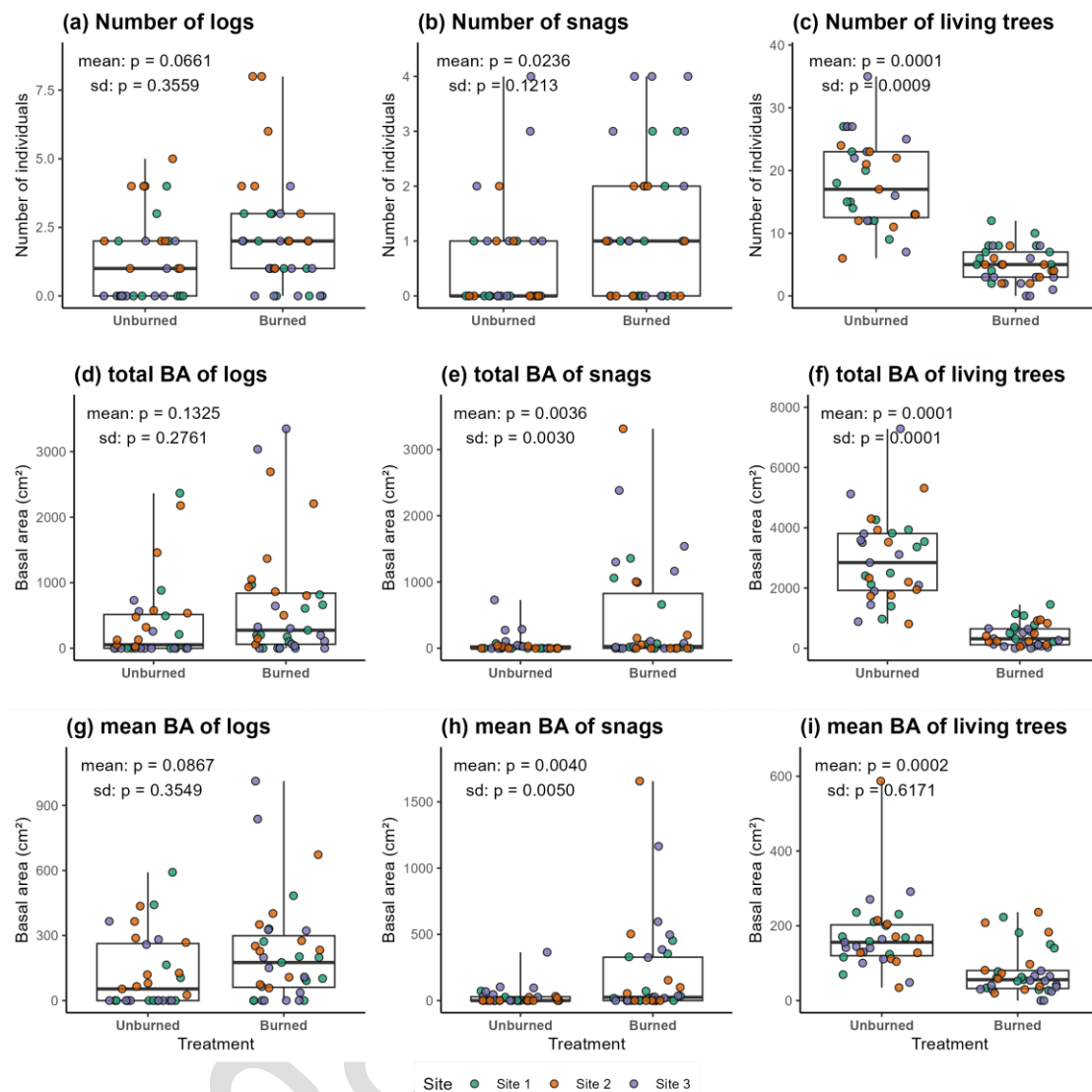


Figure 3. Structural variables of arboreal vegetation in burned and unburned forest in the Refúgio de Vida Silvestre de Una, state of Bahia, Brazil, showing the (a) number of logs (fallen dead trees), (b) number of snags (standing dead trees), (c) number of living trees, (d) total basal area of logs, (e) total basal area of snags, (f) total basal area of living trees, (g) mean basal area of logs, (h) mean basal area of snags, and (i) mean basal area of living trees. Each point represents a plot within the three studied sites. Green, orange, and purple dots indicate the first, second, and third fragments, respectively. The whiskers extend to the minimum and maximum values, the boxes represent the 25% and 75% percentiles, and the horizontal lines represent the median values. The p-values for mean and standard deviation were obtained from restricted randomization tests.

Canopy cover in the burned forests was much smaller but also much more variable than that in the unburned forests (burned mean: $58 \pm 30\%$ SD; unburned mean: $97 \pm 2\%$) (Figure 4).

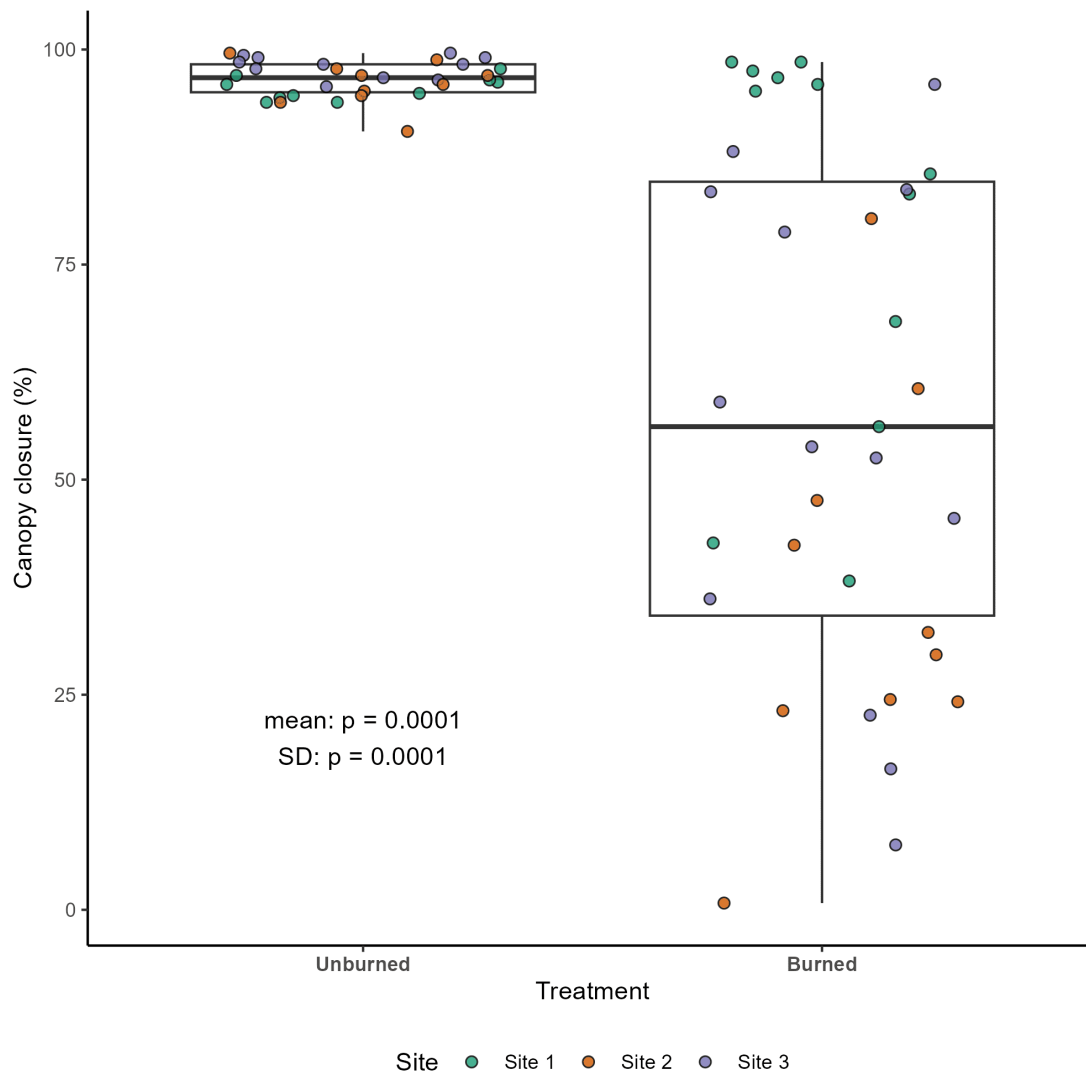


Figure 4. Variation in canopy closure between burned and unburned forests in the Refúgio de Vida Silvestre de Una, state of Bahia, Brazil. Each point represents a plot within the three studied sites (colors). Green, orange, and purple dots indicate the first, second, and third fragments, respectively. The whiskers extend to the minimum and maximum values, the boxes represent the 25% and 75% percentiles, and the horizontal lines represent the median values. The p-values for mean and standard deviation were obtained from restricted randomization tests.

Understory

There were significant differences in several response variables between burned and unburned forests. Specifically, bracken fern was more frequent in burned than in unburned forests, being present in 86% of the burned forest plots but only 3% of the unburned ones (Figure 5).

Conversely, herbaceous plants and palms were more frequent in the unburned than burned forests (Figure 5): herbaceous plants were found in 90% of the unburned forest plots but only 63% of the burned ones, whereas palms occurred in 48% of the unburned forest plots and in only 11% of the burned forest plots. Bamboos, lianas, graminoids, and saplings exhibited similar mean values between burned and unburned forests, with no statistically significant differences (Figure 5).



Figure 5. Frequency of understory cover occurrence in burned and unburned forests in the Refúgio de Vida Silvestre de Una, state of Bahia, Brazil, showing the proportion of plots with bamboos, Neotropical bracken fern (*Pteridium* sp.), graminoids, other herbaceous plants, lianas, palms, and saplings, for the three study sites. Unburned forest is in green and burned forest in dark gray. The p-values were obtained from restricted randomization tests.

DISCUSSION

Our results showed structural differences between burned and unburned forests. Variables related to mortality and basal area distribution reflect the ecological impacts of fire on forest structure, directly affecting species composition, resource availability, and natural regeneration processes. Thus, our results corroborate that forest fires are among the main disturbance agents capable of altering the structure and functioning of forest ecosystems (Perez-Quezada *et al.* 2021; Menezes *et al.* 2019; Seidl *et al.* 2017).

Although we found living trees in the burned forest, they were smaller and fewer in abundance than in the unburned forest. This likely indicates that the natural forest regeneration processes have not yet been sufficient for forest recovery. One possible explanation is the reduced input of zoochorous seeds after fire, as previously reported for burned Atlantic Forest areas (Rocha *et al.* 2021), and can be favored by the presence of remnant trees in the burned forest (Gama *et al.* 2025) as well as by the presence of *P. esculentum* (Jatobá *et al.* 2016; López *et al.* 2024).

Our results indicated a higher total basal area of living individuals in the unburned (adjacent) forests, which can contribute to the regeneration process. According to Pivello *et al.* (2006), Rocha *et al.* (2022) and Gama *et al.* (2025), seed dispersal is correlated with vegetation structure, with areas of higher tree density and unburned forest also exhibiting greater seed rain.

Although we observed a high number of snags and greater basal area in the burned forest, these differences were not statistically significant compared to the unburned forest. In addition to showing the impacts of fire, these snags may also favor forest regeneration. The presence of standing dead individuals may facilitate regeneration by acting as nucleation sites, serving as perches for disperser fauna and potentially increasing seed rain (Bellotto *et al.* 2009; Gama *et al.* 2025). Burned trees and shrubs that remain standing may serve as key structural elements during

the early stages of post-fire regeneration (Cavallero *et al.* 2013). Artificial perches have also been found to significantly enhance the input of forest seeds in pasture areas, acting as facilitators of ecological succession (Iguatemy *et al.* 2020; Zwiener *et al.* 2014).

Most fauna avoid deforested areas due to the lack of perch structures, low food resource availability, and absence of refuges (Gallegos *et al.* 2023; Gandolfi *et al.* 2009). In the northeastern Atlantic Forest, approximately 71% of tree species are dispersed by vertebrates such as birds and mammals (Silva and Tabarelli, 2000). Thus, shifts in disperser assemblages, combined with the presence of *P. esculentum*, which rapidly colonizes disturbed habitats and alters community structure, constitute major barriers to native vegetation recovery (Catterall, 2018; Costa *et al.* 2022; Jatobá *et al.* 2016).

In addition to these impacts, the increase in soil temperature following fires is often accompanied by a decrease in moisture, which can hinder vegetation development (Carvalho *et al.* 2022; Naval *et al.* 2025). On the other hand, the presence of fallen dead individuals and decomposing stems may mitigate these negative effects, contributing to organic matter accumulation and improvement of microenvironmental conditions, thus favoring seedling establishment (Sena *et al.* 2022). Even so, the canopy cover reduction in the burned forest, as observed in our study, is likely to result in a drier microclimate, creating unfavorable conditions for native forest seedlings (Carvalho *et al.* 2023; Magnago *et al.* 2015; Pivello *et al.* 2021). These effects were probably reflected in our results, which showed a low average number of regenerating individuals in burned areas.

Variables related to the occurrence of palms and herbaceous plants had higher average values in unburned forests, except for *P. esculentum*, which showed the opposite pattern, with a high occurrence in burned areas. This behavior indicates a tendency for invasive and superdominant species (Pivello *et al.* 2018) — such as *P. esculentum* and grasses — to expand under post-disturbance conditions, competing for resources and benefiting from altered

environmental conditions. The dominance of such species in burned areas has also been reported in other studies conducted in tropical forests (Carvalho *et al.* 2022; Menezes *et al.* 2019; Ross *et al.* 2023). Thus, the restoration of burned Atlantic Forest also demands the control of these species, which has been shown to be a great challenge (Xavier *et al.* 2023).

The occurrence of palms was higher in unburned forests, likely due to more favorable microclimatic conditions, which benefit species tolerant of shade and higher humidity (Colmenares-Trejos *et al.* 2024; Guimarães & Souza, 2017). Bamboo and lianas showed similar average values in both burned and unburned forests, with no significant differences. However, the rapid establishment of bamboo, due to its invasive ecology, can result in dense thickets that alter understory structure, hinder natural regeneration, and reduce local biodiversity (Lima *et al.* 2012; Carvalho *et al.* 2013; Rother *et al.* 2021). Similarly, lianas tend to benefit from disturbances and compete for resources such as light, water, and nutrients (Finlayson *et al.* 2022), compromising tree recruitment and interfering with community dynamics, thus delaying or even halting regeneration processes (Campbell *et al.* 2018).

The combined results — lower number and basal area of living trees in the burned forest, lower canopy cover, an increased occurrence of the bracken fern and a decrease in palms and herbaceous plants — indicate that the natural regeneration in these sites may not be sufficient for the forest to recover. Combined with previous results on decreases in both seed rain (Rocha *et al.* 2022) and litterfall (Rocha *et al.* 2024), in addition to severe changes in vegetation structure shortly after the fire (Menezes *et al.* 2019), this points to the necessity of active restoration strategies. These strategies would likely have to include the planting of fast-growing native trees, as has been suggested for the control of *Pteridium* sp. (Xavier *et al.* 2024), to increase canopy cover, thus improving microclimate and providing additional perching sites (Gama *et al.* 2025) for seed dispersing fauna.

Our results indicate that burned forests experienced severe impacts on forest structure and composition, with effects still evident nine years after the disturbance. The absence of clear signs of recovery suggests that these forests may remain in a prolonged state of degradation, compromising both structural and functional ecosystem attributes.

The higher basal area and structural diversity observed in unburned forests underscore the importance of adjacent forest remnants as propagule sources and regeneration nuclei. Furthermore, the results indicate that post-fire recovery is associated with structural attributes and microenvironmental conditions, suggesting that regeneration in burned forests depends not only on remnant vegetation but also on the presence of elements that support vegetation reestablishment.

Therefore, further research is needed on the effects of fire in tropical moist forests, particularly at local scales, as well as the development and implementation of ecological restoration strategies tailored to the specific conditions of affected areas, aiming to restore ecological integrity. Reconstructive restoration strategies are likely to be needed, as natural regeneration does not seem to be sufficient for effective forest regeneration.

REFERENCES

- Alves, L.F., & Metzger, J.P. 2006. Forest regeneration in secondary forest areas in the Morro Grande Forest Reserve, Cotia, SP. *Biota Neotropica* 6(2): 1–12. <https://doi.org/10.1590/S1676-06032006000200005>
- Arnold J. 2021. ggthemes: Extra Themes, Scales and Geoms for 'ggplot2'. R package version 4.2.4. <https://CRAN.R-project.org/package=ggthemes>
- Baker, S.C., Garandel, M., Deltombe, M., & Neyland, M.G. 2013. Factors influencing initial vascular plant seedling composition following either aggregated retention harvesting and

- regeneration burning or burning of unharvested forest. *Forest Ecology and Management* 306: 192–201. <https://doi.org/10.1016/j.foreco.2013.06.027>
- Balch, J.K., Massad, T.J., Brando, P.M., Nepstad, D.C., & Curran, L.M. 2013. Effects of high-frequency understory fires on woody plant regeneration in southeastern Amazonian forests. *Philosophical Transactions of the Royal Society B* 368(1619): 20120157. <https://doi.org/10.1098/rstb.2012.0157>
- Barlow, J., & Peres, C.A. 2006. Effects of single and recurrent wildfires on fruit production and large vertebrate abundance in a central Amazonian forest. *Biodiversity & Conservation* 15: 985–1012. <https://doi.org/10.1007/s10531-004-3952-1>
- Bellotto, A., Gandolfi, S., & Rodrigues, R.R. 2009. Inclusion of other life forms in the restoration process. In: Rodrigues, R.R., Brancalion, P.H.S., & Isernhagen, I. (eds.) *Atlantic Forest Restoration Pact: conceptual and practical framework for forest restoration actions*. São Paulo: LERF/ESALQ, pp. 55–61.
- Benchimol, M.; Mariano-Neto, E; Faria, D.; Rocha-Santos, L.; Souza Pessoa, M.; Gomes, F. S.; Talora, D. C.; Cazetta, E. Translating plant community responses to habitat loss into conservation practices: Forest cover matters. *Biological conservation*, v. 209, p. 499-507, 2017.
- Brasil – Instituto Chico Mendes de Conservação da Biodiversidade. 2007. *Una Wildlife Refuge*. Brasília. Available at: [https://www.gov.br/icmbio/pt-br/assuntos/biodiversidade/unidade-de-conservacao/unidades-de-biomas/mata-atlantica/lista-de- ucs/revis-de-una](https://www.gov.br/icmbio/pt-br/assuntos/biodiversidade/unidade-de-conservacao/unidades-de-biomas/mata-atlantica/lista-de-ucs/revis-de-una) (Accessed September 2025).
- Brasil – MapBiomass. 2024. Brazil had 4.48 million hectares burned between January and June 2024. MapBiomass.

- Buma, B., & Wessman, C.A. 2012. Differential species responses to compounded perturbations and implications for landscape heterogeneity and resilience. *Forest Ecology and Management* 266: 25–33. <https://doi.org/10.1016/j.foreco.2011.10.040>
- Buragohain, M.K., Dar, A.A., Babu, K.N., & Parthasarathy, N. 2007. Tree regeneration and microclimate in a liana- and bamboo-dominated semideciduous Atlantic Forest. *Forest Ecology and Management* 252(1–3): 108–117. <https://doi.org/10.1016/j.tfp.2023.100371>
- Campbell, M.J., Edwards, W., Magrach, A., Alamgir, M., Porolak, G., Mohandass, D., & Laurance, W.F. 2018. Edge disturbance drives liana abundance increase and alteration of liana–host tree interactions in tropical forest fragments. *Ecology and Evolution* 8(8): 4237–4251. <https://doi.org/10.1002/ece3.3959>
- Carvalho, A.L.D., Nelson, B.W., Bianchini, M.C., Plagnol, D., Kuplich, T.M., & Daly, D.C. 2013. Bamboo-dominated forests of the southwest Amazon: detection, spatial extent, life cycle length and flowering waves. *PLoS One* 8(1): e54852. <https://doi.org/10.1371/journal.pone.0054852>
- Carvalho, L.Z.G., Massi, K.G., Coutinho, M.P., & Magalhães, V.D. 2023. Fire effects on Atlantic Forest sites from a composition, structure and functional perspective. *Brazilian Journal of Biology* 82: e268185. <https://doi.org/10.1590/1519-6984.268185>
- Catterall, C.P., Freeman, A.N., Kanowski, J., & Freebody, K. 2012. Can active restoration of tropical rainforest rescue biodiversity? A case with bird community indicators. *Biological Conservation* 146(1): 53–61. <https://doi.org/10.1016/j.biocon.2011.10.033>
- Cavallero, L., Raffaele, E., & Aizen, M.A. 2013. Birds as mediators of passive restoration during early post-fire recovery. *Biological Conservation* 158: 342–350. <https://doi.org/10.1016/j.biocon.2012.10.004>

- Christo, A.v.D.; Martins, S.V.; Balestrin, D.; Torres, FTP. 2021. Fire effect on the natural regeneration and litter accumulation in Atlantic Forest. *Scientia Forestalis* 49: e3501. <https://doi.org/10.18671/scifor.v49n130.10>
- Colmenares-Trejos, S.L., Portela, R.D.C.Q., Fernandez, A.P., Braz, M.I.G., & de Mattos, E.A. 2024. Light and dispersal strategies structure palm community along an elevation gradient in the Atlantic Forest. *Brazilian Journal of Botany* 47(2): 275–293. <https://doi.org/10.1007/s40415-024-00>
- Costa, A., Heleno, R., Dufrene, Y., Huckle, E., Gabriel, R., Doudee, D., & Kaiser-Bunbury, C.N. 2022. Seed dispersal by frugivores from forest remnants promotes the regeneration of adjacent invaded forests in an oceanic island. *Restoration Ecology* 30(8): e13654. <https://doi.org/10.1111/rec.13654>
- Cury, R.T.D.S., Montibeller-Santos, C., Balch, J.K., Brando, P.M., & Torezan, J.M.D. 2020. Effects of fire frequency on seed sources and regeneration in southeastern Amazonia. *Frontiers in Forests and Global Change* 3: 82. <https://doi.org/10.3389/ffgc.2020.00082>
- De Andrade, R.B., Balch, J.K., Carreira, J.Y., Brando, P.M., & Freitas, A.V. 2017. The impacts of recurrent fires on diversity of fruit-feeding butterflies in a southeastern Amazon forest. *Journal of Tropical Ecology* 33(1): 22–32. <https://doi.org/10.1017/S0266467416000559>
- Dodonov, P., Menezes, G.S.C., Caitano, B., Cazetta, E., & Mielke, M.S. 2019. Air and soil temperature across fire-created edges in a Neotropical rainforest. *Agricultural and Forest Meteorology* 276: 107606. <https://doi.org/10.1016/j.agrformet.2019.06.005>
- Driscoll, D.A., Armenteras, D., Bennett, A.F., Brotons, L., Clarke, M.F., Doherty, T.S., ... & Wevill, T. 2021. How fire interacts with habitat loss and fragmentation. *Biological Reviews* 96: 976–998. <https://doi.org/10.1111/brv.12687>

- Eichenwald, A.J., Grady, J.M., Knott, J.A., Read, Q.D., Rodriguez, J.M., & Record, S. 2025. The impact of disturbance on tree size distributions in the United States. *Global Ecology and Biogeography* 34: e70102. <https://doi.org/10.1111/geb.70102>
- Finlayson, C., Roopsind, A., Griscom, B.W., Edwards, D.P., & Freckleton, R.P. 2022. Removing climbers more than doubles tree growth and biomass in degraded tropical forests. *Ecology and Evolution* 12(3): e8758. <https://doi.org/10.1002/ece3.8758>
- Fisher, J.L., Loneragan, W.A., Dixon, K., Delaney, J., & Veneklaas, E.J. 2009. Altered vegetation structure and composition linked to fire frequency and plant invasion in a biodiverse woodland. *Biological Conservation* 142(10): 2270–2281. <https://doi.org/10.1016/j.biocon.2009.05.001>
- Gallegos, S.C., Mayta, C., Villegas, M., Ayala, G.M., Naoki, K., Rechberger, J., ... & Hensen, I. 2024. Habitat differences in seed-dispersing vertebrates indicate dispersal limitation in tropical bracken-dominated deforested areas. *Biotropica* 56(3): 1–10. <https://doi.org/10.1111/btp.13317>
- Gama, M., Dodonov, P., & Cazetta, E. 2025. Remnant trees increase seed rain in burned Neotropical forests. *Forest Ecology and Management* 592: 122794. <https://doi.org/10.1016/j.foreco.2025.122794>
- Gandolfi, S., Belotto, A., & Rodrigues, R.R. 2009. Incorporation of the functional group concept into restoration based on species biology. In: Rodrigues, R.R., Brancalion, P.H.S., & Isernhagen, I. (eds) *Atlantic Forest Restoration Pact: conceptual and practical framework for forest restoration actions*. São Paulo: Instituto BioAtlântica; LERF/ESALQ.
- Guimarães, L.A.O.P., & Souza, R.G. 2017. *Juçara palm: a natural heritage of the Atlantic Forest in Espírito Santo*. Vitória: Incaper.

- Hidasi-Neto, J., Barlow, J., & Cianciaruso, M.V. 2012. Bird functional diversity and wildfires in the Amazon: the role of forest structure. *Animal Conservation* 15(4): 407–415. <https://doi.org/10.1111/j.1469-1795.2012.00528.x>
- Iguatemy, M.D.A., Vilarinhos, J.A., Oda, G.A.M., Conde, M.D.M.S., & Zaú, A.S. 2020. Artificial perches: Ecological and functional aspects of their contribution in the Atlantic Forest. *Floresta e Ambiente* 27(2): e20180301. <https://doi.org/10.1590/2179-8087.030118>
- Jatoba, L. J.; Varela, R. M.; Molinillo, J. M. G.; Ud Din, Z.; Juliano Gualtieri, S. C.; Rodrigues-Filho, E.; Macías, F. A. Allelopathy of bracken fern *Pteridium arachnoideum*: new evidence from green fronds, litter, and soil. *PLoS One*, v. 11, n. 8, p. e0161670, 2016.
- Johnstone, J.F., Allen, C.D., Franklin, J.F., Frelich, L.E., Harvey, B.J., Higuera, P.E., Mack, M.C., Meentemeyer, R.K., Metz, M.R., Perry, G.L.W., Schoennagel, T., & Turner, M.G. 2016. Changing disturbance regimes, ecological memory, and forest resilience. *Frontiers in Ecology and the Environment* 14(7): 369–378. <https://doi.org/10.1002/fee.1311>
- Kassambara, A. 2025. ggpubr: 'ggplot2' Based Publication Ready Plots. R package version 0.6.1. <https://CRAN.R-project.org/package=ggpubr>
- López, C. L.; Mayta, C.; Fuentes, A. F.; Villegas, M.; Jimenez, E. A.; Vasquez, V.; Hensen, I.; Gallegos, S. C. Disentangling the roles of bracken fronds and litter on natural seedling recruitment in fire-disturbed tropical montane habitats. *Forest Ecology and Management*, v. 566, p. 122056, 2024.
- Lima, R.A., Rother, D.C., Muler, A.E., Lepsch, I.F., & Rodrigues, R.R. 2012. Bamboo overabundance alters forest structure and dynamics in the Atlantic Forest hotspot. *Biological Conservation* 147(1): 32–39. <https://doi.org/10.1016/j.biocon.2012.01.015>
- Magnago, L.F.S., Rocha, M.F., Meyer, L., Martins, S.V., & Meira-Neto, J.A.A. 2015. Microclimatic conditions at forest edges have significant impacts on vegetation structure

- in large Atlantic Forest fragments. *Biodiversity and Conservation* 24(9): 2305–2318.
<https://doi.org/10.1007/s10531-015-0961-1>
- Malhi, Y. 2012. The productivity, metabolism and carbon cycle of tropical forest vegetation. *Journal of Ecology* 100(1): 65–75. <https://doi.org/10.1111/j.1365-2745.2011.01916.x>
- Malhi, Y., Gardner, T.A., Goldsmith, G.R., Silman, M.R., & Zelazowski, P. 2014. Tropical forests in the Anthropocene. *Annual Review of Environment and Resources* 39(1): 125–159.
<https://doi.org/10.1146/annurev-environ-030713-155141>
- Manly, B.J.F. 2007. Randomization, bootstrap and Monte Carlo methods in biology. 3rd ed. Boca Raton: Chapman & Hall/CRC. <https://doi.org/10.1201/9781315273075>
- Menezes, G.S.C., Cazetta, E., & Dodonov, P. 2019. Vegetation structure across fire edges in a Neotropical rainforest. *Forest Ecology and Management* 453: 117587.
<https://doi.org/10.1016/j.foreco.2019.117587>
- Metzger, J.P., Martensen, A.C., Dixo, M., Bernacci, L.C., Ribeiro, M.C., Teixeira, A.M.G., & Pardini, R. 2009. Time-lag in biological responses to landscape changes in a highly dynamic Atlantic Forest region. *Biological Conservation* 142(6): 1166–1177.
<https://doi.org/10.1016/j.biocon.2009.01.033>
- Naval, M.L.M., Bieluczyk, W., Alvarez, F., da Silva Carvalho, L.C., Maracahipes-Santos, L., de Oliveira, E.A., ... & Feldpausch, T.R. 2025. Impacts of repeated forest fires and agriculture on soil organic matter and health in southern Amazonia. *Catena* 254: 105123.
- Paz, H., Ortiz-Alcaraz, A., & del-Val, E. 2022. The effects of the aggressive species *Pteridium caudatum* on the vegetation of Socorro Island: Restoration challenges and opportunities. *Journal for Nature Conservation* 67: 126160. <https://doi.org/10.1016/j.jnc.2022.126160>
- Perez-Quezada, J.F., Urrutia, P., Olivares-Rojas, J., Mejjide, A., Sánchez-Cañete, E.P., & Gaxiola, A. 2021. Long-term effects of fire on the soil greenhouse gas balance of an old-

- growth temperate rainforest. *Science of the Total Environment* 755: 142442.
<https://doi.org/10.1016/j.scitotenv.2020.142442>
- Pimm, S.L., & Raven, P. 2000. Extinction by numbers. *Nature* 403(6772): 843–845.
- Pivello, V.R., Petenon, D., Jesus, F.M.D., Meirelles, S.T., Vidal, M.M., Alonso, R.D.A.S., Franco, G.A.D.C., & Metzger, J.P. 2006. Seed rain in Atlantic Forest fragments (São Paulo State, SP, Brazil) with different connectivity, forest structure and distance to edge. *Acta Botanica Brasilica* 20: 845–859. <https://doi.org/10.1590/S0102-33062006000400010>
- Pivello, V.R., Vieira, I., Christianini, A.V., Ribeiro, D.B., da Silva Menezes, L., Berlinck, C.N., Melo, F.P.L., Marengo, J.A., Tornquist, C.G., Tomas, W.M., & Overbeck, G.E. 2014. Contribution of artificial perches to seed dispersal and its application in forest restoration. *Ciência Florestal* 24: 501–507. <https://doi.org/10.1016/j.pecon.2021.06.005>
- Pivello, V.R., Vieira, I., Christianini, A.V., Ribeiro, D.B., da Silva Menezes, L., Berlinck, C.N., Melo, F.P.L., Marengo, J.A., Tornquist, C.G., Tomas, W.M., & Overbeck, G.E. 2021. Understanding Brazil's catastrophic fires: causes, consequences and policy needed to prevent future tragedies. *Perspectives in Ecology and Conservation* 19(3): 233–255. <https://doi.org/10.1016/j.pecon.2021.06.005>
- Pivello, V.R., Vieira, M.V., Grombone-Guaratini, M.T., & Silva-Matos, D.M. 2018. Thinking about super-dominant populations of native species — examples from Brazil. *Perspectives in Ecology and Conservation* 16: 74–82.
- R Core Team. 2022. R: A language and environment for statistical computing. Vienna: R Foundation for Statistical Computing. URL <https://www.R-project.org/>
- Ribeiro, M.C., Metzger, J.P., Martensen, A.C., Ponzoni, F.J., & Hirota, M.M. 2009. The Brazilian Atlantic Forest: how much is left, and how is the remaining forest distributed? Implications for conservation. *Biological Conservation* 142(6): 1141–1153. <https://doi.org/10.1016/j.biocon.2009.02.021>

- Rocha, J.I.S., Menezes, G.S.C., Cazetta, E., Dodonov, P., & Talora, D.C. 2022. Seed rain across fire-created edges in a Neotropical rainforest. *Plant Ecology* 223(3): 247–261. <https://doi.org/10.1007/s11258-021-01205-6>
- Rocha, J.I.S., Talora, D.C., & Dodonov, P. 2024. Effects of fire and edges on plant litterfall in a Neotropical rainforest, Brazil. *Plant Ecology* 225(4): 317–330. <https://doi.org/10.1007/s11258-024-01415-8>
- Rocha-Santos, L.; Benchimol, M.; Mayfield, M. M.; Faria, D.; Pessoa, M. S.; Talora, D. C.; Mariano-Neto, E.; Cazetta, E. Functional decay in tree community within tropical fragmented landscapes: Effects of landscape-scale forest cover. *PLoS One*, v. 12, n. 4, p. e0175545, 2017.
- Ross, T., Srivastava, S.K., & Shapcott, A. 2023. Investigating the relationship between fire severity and post-fire vegetation regeneration and subsequent fire vulnerability. *Forests* 14(2): 222. <https://doi.org/10.3390/f14020222>
- Rother, D.C., Jordano, P., Rodrigues, R.R., & Pizo, M.A. 2013. Demographic bottlenecks in tropical plant regeneration: a comparative analysis of causal influences. *Perspectives in Plant Ecology, Evolution and Systematics* 15(2): 86–96. <https://doi.org/10.1016/j.ppees.2012.12.004>
- Sansevero, J.B., Garbin, M.L., Sánchez-Tapia, A., Valladares, F., & Scarano, F.R. 2020. Fire drives abandoned pastures to a savanna-like state in the Brazilian Atlantic Forest. *Perspectives in Ecology and Conservation* 18(1): 31–36. <https://doi.org/10.1016/j.pecon.2019.12.004>
- Sansevero, J.B.B., Prieto, P.V., Sanchez-Tapia, A., Braga, J.M.A., & Rodrigues, P.J.F.P. 2017. Past land-use and ecological resilience in a lowland Brazilian Atlantic Forest: implications for passive restoration. *New Forests* 48(5): 573–586. <https://doi.org/10.1007/s11056-017-9586-4>

- Scheper, A.C., Verweij, P.A., & Van Kuijk, M. 2021. Post-fire forest restoration in the humid tropics: a synthesis of available strategies and knowledge gaps for effective restoration. *Science of the Total Environment* 771: 144647. <https://doi.org/10.1016/j.scitotenv.2020.144647>
- Seidl, R., Thom, D., Kautz, M., Martin-Benito, D., Peltoniemi, M., Vacchiano, G., Wild, J., Ascoli, D., Petr, M., Honkaniemi, J., Lexer, M.J., Trotsiuk, V., Mairota, P., Svoboda, M., Fabrika, M., Nagel, T.A., & Reyer, C.P.O. 2017. Forest disturbances under climate change. *Nature Climate Change* 7(6): 395–402. <https://doi.org/10.1038/nclimate3303>
- Sena, P.H.A., Fonsêca, N.C., & Lins-e-Silva, A.C.B. 2022. Non-negligible role of dead organic matter in a rainforest remnant in Northeast Brazil. *Rodriguésia* 73. <https://doi.org/10.1590/2175-7860202273041>
- Silva, J.M.C., & Tabarelli, M. 2000. Tree species impoverishment and the future flora of the Atlantic Forest of northeast Brazil. *Nature* 404: 72–74. <https://doi.org/10.1038/35003563>
- Silva, I.A., Valenti, M.W., & Silva-Matos, D.M. 2009. Fire effects on the population structure of *Zanthoxylum rhoifolium* Lam (Rutaceae) in a Brazilian savanna. *Brazilian Journal of Biology* 69: 813-818.
- Silva, S.S., Fearnside, P.M., de Alencastro Graça, P.M.L., Numata, I., de Melo, A.W.F., Ferreira, E.L., ... & de Lima, P. R. F. (2021). Increasing bamboo dominance in southwestern Amazon forests following intensification of drought-mediated fires. *Forest Ecology and Management*, 490: 119139. [10.1016/j.foreco.2021.119139](https://doi.org/10.1016/j.foreco.2021.119139)
- Tabarelli, M., Pinto, L.P., Silva, J.M., Hirota, M., & Bede, L. 2005. Challenges and opportunities for biodiversity conservation in the Brazilian Atlantic Forest. *Conservation Biology* 19(3): 695–700. <https://doi.org/10.1111/j.1523-1739.2005.00694.x>

- Trejo, M.D.R.R., Pérez-García, B., Pérez-Salicrup, D.R., & Orozco-Segovia, A. 2010. Effect of fire on the germination of spores of *Pteridium caudatum*, an invasive fern. *Journal of Tropical Ecology* 26(4): 457-465. 10.1017/S0266467410000118
- Vancine, M.H., Muylaert, R.L., Niebuhr, B.B., de Faria Oshima, J.E., Tonetti, V., Bernardo, R., Angelo, C., Rosa, M.R., Grohmann, C.H., & Ribeiro, M.C. 2024. The Atlantic Forest of South America: Spatiotemporal dynamics of the vegetation and implications for conservation. *Biological Conservation* 291: 110499. <https://doi.org/10.1016/j.biocon.2024.110499>
- Wickham, H. 2016. *ggplot2: Elegant Graphics for Data Analysis*. Springer-Verlag, New York.
- Wickham, H., & Pedersen, T. 2019. *gtable: Arrange 'Grobs' in Tables*. R package version 0.3.0. <https://CRAN.R-project.org/package=gtable>
- Wickham, H., Averick, M., Bryan, J., Chang, W., McGowan, L.D., François, R., Grolemund, G., Hayes, A., Henry, L., Hester, J., Kuhn, M., Pedersen, T.L., Miller, E., Bache, S.M., Müller, K., Ooms, J., Robinson, D., Seidel, D.P., Spinu, V., Takahashi, K., Vaughan, D., Wilke, C., Woo, K., & Yutani, H. 2019. Welcome to the tidyverse. *Journal of Open Source Software* 4(43): 1686. <https://doi.org/10.21105/joss.01686>
- Wickham, H., Pedersen, T., & Seidel, D. 2023. *scales: Scale functions for visualization*. R package version 1.3.0. Available at: <https://CRAN.R-project.org/package=scales>
- Wilke, C. 2024. *cowplot: Streamlined Plot Theme and Plot Annotations for 'ggplot2'*. R package version 1.1.3. <https://CRAN.R-project.org/package=cowplot>
- Xavier, R.O., Melo, U.M., Pivello, V.R., Marrs, R.H., Castro, P.G.A., Nascimento, J.L., & Silva-Matos, D.M. 2023. Combining mechanical control and tree planting to restore montane Atlantic forests dominated by the Neotropical bracken (*Pteridium arachnoideum*). *Forest Ecology and Management* 529: 120657.

Yu, G. 2025. ggimage: Use Image in 'ggplot2'. R package version 0.3.4. <https://CRAN.R-project.org/package=ggimage>

Zar, J.H. 2010. Biostatistical Analysis. 5th ed. Upper Saddle River: Pearson.

Zwiener, V.P., Cardoso, F.C., Padial, A.A., & Marques, M.C. 2014. Disentangling the effects of facilitation on restoration of the Atlantic Forest. Basic and Applied Ecology 15: 34–42. <https://doi.org/10.1016/j.baae.2013.11.005>

Submitted: 13 November 2025

Accepted: 3 August 2026

Published online: 06 September 2026

Associate Editor: Dr. Edson Gomes de Moura-Júnior