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Effect of Prescribed Burning on Tree Diversity, Biomass Stocks and Soil Organic Carbon Storage in Tropical Highland Forests

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Abstract: Fire has been an integral part of ecosystem functioning in many biomes for a long time, but the increased intensity and frequency of wildfires often affect plant diversity and carbon storage. Prescribed burning is one of the alternatives to forest fuel management where the fire is controlled and carried out under a determined set of weather conditions and objectives. The effect of prescribed burning on plant diversity and carbon (C) storage has not been studied widely. The objective of this study was to evaluate the effect of prescribed burning on plant diversity indices, biomass stocks, and soil C storage in the tropical highland forests of Southern Mexico. We assessed plant diversity and carbon stocks at 21 sampling sites: seven with prescribed burning, seven non-burning, and seven with wildfires. We calculated tree biodiversity indices, stand structural properties, and species composition among burning treatments. We quantified C stocks in vegetation biomass by using an allometric equation and forest litter by direct sampling. We analyzed 252 soil samples for soil organic C content and other properties. The results showed that the biodiversity index was higher in sites with prescribed burning (Shannon index, $H = 1.26$) and non-burning ($H = 1.14$) than in wildfire sites ($H = 0.36$). There was a greater similarity in plant species composition between non-burning and prescribed burning sites compared to wildfire sites. Prescribed burning showed a positive effect on soil carbon storage (183.9 Mg C ha^{-1}) when compared to wildfire (144.3 Mg C ha^{-1}), but the difference was not statistically significant ($p > 0.05$) in biomass stocks. Prescribed burning in this study conserved plant diversity as well as soil carbon stocks compared to non-burning, the opposite of what we found in wildfires.

Keywords: carbon stocks; controlled forest fire; tree species diversity; forest regeneration; wildfire

1. Introduction

Carbon storage in forest ecosystems depends on multiple natural and anthropogenic factors [1–3]. Fire is one such factor that affects carbon storage in forest ecosystems [4,5]. Although fire has been used for thousands of years and plays an important role in regulating ecosystem functions, anthropogenic global climate change has increased the frequency and intensity of mega fires throughout the world [6,7]. Alterations in local weather patterns and global warming have a significant impact on the outbreak, scale, and intensity of wildfires [8–10]. Globally, around 348 million hectares of land are burned per year, which emits about 2.2 Pg of carbon into the atmosphere [11–14]. Tropical forests account for about 15% of the global burnt area [12].

Mexico is susceptible to wildfires because of its climate, relief, vegetation, and soil types [15,16]. Every year, about 21,000 fire events affect nearly 677,000 ha of forest area in Mexico and Central America [13,17,18]. For example, from January to December 2020, there were records of 5913 wildfires affecting 378,928 hectares of forests [18].

Fire effects on biodiversity are the results of interaction between multiple ecosystem processes during and after burning [5,19]. Wildfire influences forest biodiversity due to the changes in seed germination, seedling establishment, tree mortality, and post-fire competition between native and alien species [20,21]. In some ecosystems, prolonged drought followed by a high-intensity wildfire can reduce up to 20% of the biomass stock due to the loss of vegetation cover [22,23]. Burning initially reduces litter and deadwood mass accumulated on the ground surface but it contributes to mineralization and nutrient release from organic matter burning.

Fire induces changes in soil organic carbon storage and soil physicochemical properties [24,25]. Some studies stated that high-intensity wildfire reduces the population of soil-dwelling biota, directly affecting organic matter reserve, soil moisture, and temperature, thus altering nutrient cycling and other biogeochemical processes [26,27]. Depending on the severity, burning can modify the proportions of the labile and recalcitrant organic matter in the soil and the rate of organic matter decomposition [28]. Such changes in vegetation dynamics and soil conditions have a direct impact on post-fire recovery of forest ecosystems and carbon storage in above- and belowground pools [29]. The type and severity of burning can have a differential impact on vegetation succession and carbon recovery pathways. Some studies reported that high-severity wildfires lead to a greater loss of understory plant cover and trees compared to low-severity wildfires [30,31].

Prescribed burning (PB) is carried out for different purposes where the fire is controlled and carried out under certain selected weather conditions [24,32]. It is used to reduce the severity or to alter the behavior of wildfires in many parts of the globe. For example, managing excessive forest fuel through PB can help reduce the outbreak, scale, and severity of wildfires [33]. It is also used as a management tool for fuel reduction and the creation of seedbeds in ecosystems adapted to fire [24,32,34]. Some studies reported that PB has minimal effects on the crowns of adult trees and facilitates seedling recruitment, thus helping to restore biodiversity [35–37]. Avoidance of high-intensity fire risks through PB also contributes to sparing CO_2 emissions from possible large-scale biomass burning [38,39].

However, studies on the effect of prescribed burning on plant biodiversity indices and carbon storage in above- and belowground pools are still limited in many parts of the world. Furthermore, the results regarding the effect of prescribed burning on forest biomass and soil carbon stocks are controversial. Some studies reported that prescribed burning has a positive or neutral effect on carbon sequestration [40–42], while others reported a negative effect leading to carbon loss [22,43,44]. While the damage caused by wildfires is more frequent in southern Mexico, there is an increasing need to study ecosystem response to prescribed burning in terms of plant diversity and carbon storage. Therefore, the objective of this study was to evaluate the effect of prescribed burning on plant diversity indices, forest biomass stocks, and soil carbon storage in tropical highland forest ecosystems in southern Mexico.

2. Materials and Methods

2.1. Study Sites

The research was carried out in tropical highland forests distributed within two municipalities of Chiapas, México: Villaflores and Villa Corzo, both in the Frailesca region (Figure 1). These forests are submerged within the tropical region of southern Mexico but are distributed in the mountainous landscape with elevations higher than 900 m above sea level. The area is a part of the Sierra Madre de Chiapas. Histories of fire have registered in some portions of these forests, but the frequency, scale, and severity have been different with time [45]. The climate in the region is warm sub-humid, with annual rainfall between 1300 and 2000 mm and an average temperature of 24 °C. Study sites were located at elevation from 838 to 1520 m above mean sea level. Sampling was carried out in tropical highland forests (Figure 2) with *Pinus oocarpa*, *Quercus peduncularis*, and *Byrsonima crassifolia* as the most abundant species [46]. Leptosol and rendzina are the most common soil types found in the study sites. These are relatively shallow soils of 30–40 cm depth with a moderate amount of organic matter. Soils in the region are moderately acidic with pH ranging from 4.0 to 5.0 and are characterized by clay loam to silty clay loam texture [47].

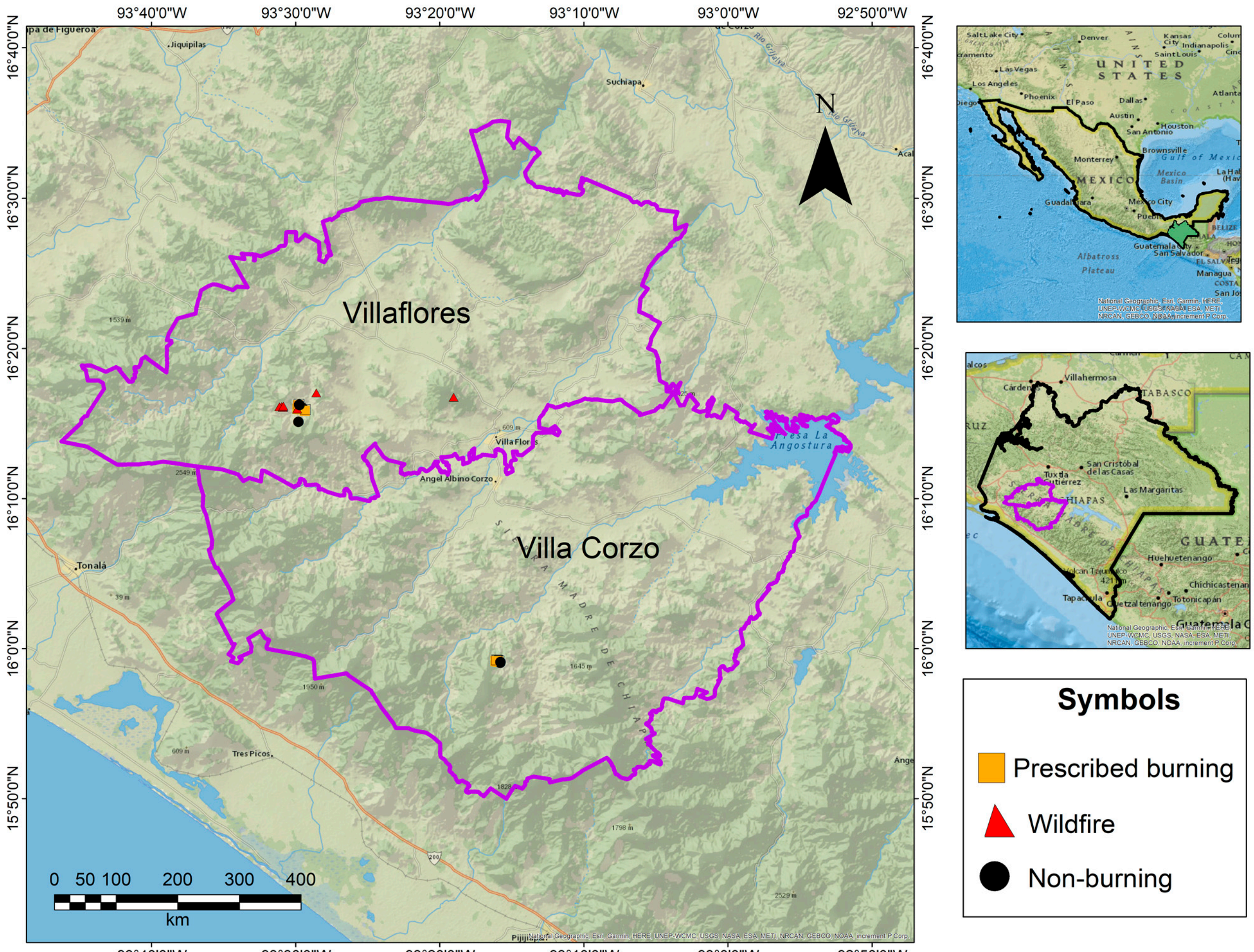

Figure 1. Location of the study area and distribution of sampling sites in two municipalities of Chiapas, southern Mexico. The area is a part of the Sierra Madre de Chiapas.

Prescribed burning is the intentional use of fire by management actions under applicable regulations to meet specific objectives, while wildfire is an unplanned wildland fire or blaze caused naturally or by humans without authorization [48]. In this study, prescribed burns were carried out in 2015 and 2016, following the Mexican government's NOM-015 rules and guidelines which establish the technical specifications for the use of controlled fire in forests, crops, and rangeland ecosystems [49]. It was carried out under constant monitoring of wind speed and direction, atmospheric temperature, and relative humidity by trained personnel [15]. The blaze of 2015 affected the sites we sampled as wildfire, whereas non-burning sites had no history of fire for the past 30 years. In prescribed burning sites, the fire was intentionally used to reduce forest fuel load by the end of the winter season but before the start of the hot dry season. It was carried out in the morning hour when the air temperature was about 18–25 °C, the atmospheric humidity was higher than

40%, and the wind speed was low. Ignition started from the upper part of the slopy terrain and proceeded downward. The perimeter of each destined area was cleared for firebreaks before ignition. The thickness of forest fuel ranged between 5 and 12 cm before burning (Figure 2).

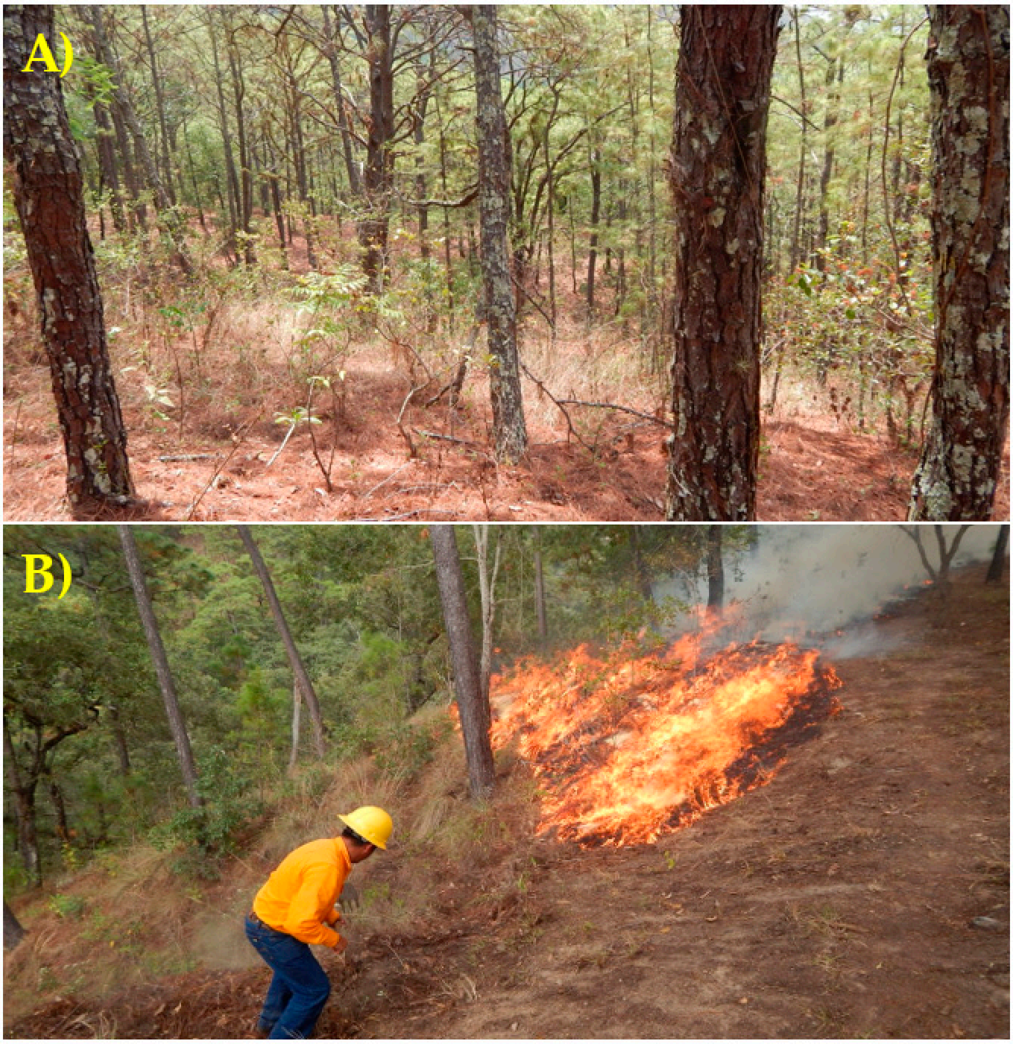

Figure 2. Images of forest ecosystems representing vegetation type (**A**) and prescribed burning process (**B**). Photos were taken from study sites located in Chiapas, Mexico. (Photo: Alonso López-Cruz.)

2.2. Sampling and Measurement of Trees

Areas of non-burning, prescribed burning, and wildfire were selected based on vegetation, topography, and soil characteristics. Prescribed burning was carried out strategically in areas without fire history for at least the past 30 years but adjacent to the areas affected by wildfire. On average, 10 hectares of land was prescribed burned from each of the seven sites. In each site, a circular 1000 m^2 plot was established within the 10-hectare burning frame. For all treatments (prescribed burning, non-burning, and wildfire), sampling was carried out leaving about a 50 m distance from the edge to avoid the border effect of other fire treatments. Sampling was carried out in the years 2020 and 2021, about five years after burning disturbance. Sampling consisted of 21 circular plots (7 with prescribed burning, 7 non-burning, and 7 wildfires), each with a 1000 m^2 area. We used the inner 400 m^2 nested plot for trees smaller than 7.5 cm diameter at breast height (DBH, 1.30 m above ground level) and the 1000 m^2 plot for all trees equal to or greater than 7.5 cm DBH. To carry out the inventory process systematically, we divided the plots into four quadrants using the cardinal points as references. We measured tree DBH, crown diameter, and tree height by using diameter tapes, meter tapes, and clinometers.

2.3. Diversity and Ecological Indices

With the tree inventory data, we calculated biodiversity and other ecological indices. For the Shannon–Weaver diversity index, we used the relation between the number of species and the number of individuals that make up each species (Equation (1)). This index increases when the number of individuals of each species is similar. Simpson's dominance index (Equation (2)) has a value between 0 and 1, with 1 being the value with the least presence of species richness. When there are a few species dominating with many individuals and other species with only a few individuals, the D index nears 1.

$$Shannons\ H = -\sum_{i}^{s} pi \log pi \tag{1}$$

$$Simsons\ D = \sum_{i}^{S} pi^2 \tag{2}$$

where:

H: Shannon–Weaver diversity index;
D: Simpson dominance index;
S: Number of species or species richness;
pi: Proportion of the total sample found for the species i.

The relative importance of the species was calculated as the sum of the relative frequency (Equation (3)), relative dominance (Equation (4)), and relative abundance (Equation (5)) [50]. For dominance, we used the sum of the basal area of the tree trunks. Frequency indicated the number of plots that the species was inventoried in, while for abundance, we used tree counts per species.

$$Relative\ frequency = \frac{Frequency\ of\ the\ species}{Frequency\ of\ all\ the\ species} \times 100 \tag{3}$$

$$elative\ dominance = \frac{Basal\ area\ of\ the\ species}{Basal\ area\ of\ all\ the\ species} \times 100 \tag{4}$$

$$Relative\ abundance = \frac{Number\ of\ individuals\ of\ the\ species}{Total\ number\ of\ individual\ from\ all\ the\ species} \times 100 \tag{5}$$

To assess the similarity/dissimilarity of species composition between sites with prescribed burning, non-burning, and wildfires, we used Sorensen's coefficient (Equation (6)) and Jaccard's index (Equation (7)) [51].

$$SCS = \frac{2c}{a+b} \times 100 \tag{6}$$

$$JSI = \frac{c}{a+b-c} \times 100 \tag{7}$$

where:

SCS: Sorensen's coefficient of similarity;
JSI: Jaccard's similarity index;
a: Number of species occurring in the community A;
b: Number of species occurring in community B;
c: Number of species occurring in both communities.

2.4. Biomass Estimation

We estimated the aboveground living biomass (AGB) of each tree using published allometric equations with DBH and the total tree height data. For *Pinus oocarpa* and *Quercus* *peduncularis*, we used species-specific equations [52], and for other tropical plant species, we used a general equation for tropical trees [53]. We calculated the coarse root biomass by using the AGB-dependent equation [54].

We collected forest floor litter samples using four 30 × 30 cm square frames from each nested plot. Fresh and fermented litter samples were collected separately. The samples were oven-dried and analyzed from their carbon concentrations. To sample fallen deadwood material, we used the planar intersection method [55]. We converted the volume of the deadwood to its mass by using decay phase-specific wood density values [56].

2.5. Soil Sampling and Analysis

From every plot, we collected four random soil samples separately for each of the depth classes: 0–10, 10–20, and 20–30 cm for SOC analysis. Another four samples were collected parallelly for bulk density estimation. This way, we sampled a total of 504 soil samples, with 252 for chemical analysis and 252 for bulk density. We used the 10 cm long and 5.3 cm diameter steel cylinders to collect soil samples. We analyzed the carbon content (%) of each sample using the chemical digestion method followed by a sucrose-calibrated spectrophotometer (GENESYS 10SUV-Vis, Thermos Fisher Scientific Inc., Waltham, Massachusetts, USA) reading at 600 nm [57,58]. We calculated bulk density by oven-drying the volume-known soil samples at 105 °C for 72 h. We also analyzed soil pH and redox potential (Eh) by using a potentiometer. Before analysis, we sieved the soil samples (using 2 mm mesh) and removed any coarse fragments. Fine root biomass was separated from the soil, cleaned, oven-dried, and weighed [59].

2.6. Data Analysis

We checked all the data for the normality of distribution using the Shapiro–Wilk test. Non-normal data were log-transformed before parametric tests. To test the significant difference between burning types, we used one-way ANOVA. To analyze the effect of burning types, soil depth, and the interaction between them on SOC and soil properties, we used factorial ANOVA. Tukey's HSD test ($p < 0.05$) was used to compare mean differences between sites with prescribed burning, non-burning, and wildfires. Descriptive statistics such as mean, standard error, and confidence intervals were back-transformed where necessary. To test the associations between variables such as tree biomass, SOC content, soil properties, and topography, we used a principal component analysis (PCA) based on correlations. We applied the normalized varimax rotation for PCA and considered eigenvalues greater than one to extract the principal components. Variables with correlations higher than 0.5 with principal components were extracted. We used SPSS software for this analysis [60].

3. Results

3.1. Ecological Indices

Prescribed burning sites showed a Shannon index (H) of 1.26, statistically similar to non-burning sites (H = 1.14) but significantly higher than wildfire sites (H = 0.36) (Table 1). Simpson's dominance index was the highest in wildfire sites (D = 0.67) and the lowest in prescribed burning (Table 2). This indicates that the sites with wildfires were dominated by a few species, making them less diverse. Regarding stand structural parameters, we did not find statistically significant differences between burning types, but the average tree height, crown diameter, and basal area tended to decrease in wildfire sites compared to non-burning and prescribed burning sites (Table 1). For example, the basal area at prescribed burning sites was 12.34 m^2 ha^{-1} compared to 9.97 m^2 ha^{-1} at wildfire sites.

Table 1. Tree diversity indices and forest structural parameter differences between prescribed burning, non-burning, and wildfire sites of tropical highland forests in Chiapas, Mexico. Different superscript letters followed by mean values denote significant differences between burning treatments (Tukey HSD $p < 0.05$).

Diversity and Structure	DBH Category	Mean (95% Confidence Interval)		
		Prescribed Burning	Non-Burning	Wildfire
Shannon (*H*) index	All trees	1.26 (0.97–1.55) [a]	1.14 (0.81–1.47) [a]	0.36 (0.12–0.59) [b]
Simpson (*D*) index	All tress	0.37 (0.22–0.52) [a]	0.42 (0.28–0.56) [a]	0.67 (0.37–0.97) [a]
Tree height (m)	<7.5 cm	1.73 (0.73–4.10)	5.95 (3.85–9.19)	3.59 (1.51–8.51)
	>7.5 cm	13.50 (11.72–15.56)	12.99 (11.25–15.00)	11.76 (10.19–13.56)
	All trees	13.00 (11.79–14.33)	11.42 (9.63–13.56)	10.73 (8.76–13.14)
Crown diameter (m)	<7.5 cm	1.76 (0.72–4.29)	5.69 (3.31–9.81)	1.58 (0.71–3.51)
	>7.5 cm	53.25 (25.31–112.05)	37.89 (15.59–92.07)	24.30 (11.66–50.62)
	All trees	50.80 (23.13–111.59)	31.52 (13.45–73.88)	20.90 (9.83–44.42)
Basal area (m^2 ha^{-1})	<7.5 cm	0.81 (0.57–1.16)	0.21 (0.10–0.43)	0.28 (0.10–0.76)
	>7.5 cm	12.22 (6.73–22.17)	10.88 (7.68–15.43)	9.77 (7.15–13.35)
	All trees	12.34 (6.78–22.45)	11.12 (7.81–15.84)	9.97 (7.41–13.41)

Ten tree species were identified from all sites, but based on the relative importance value index (RVI), three were considered predominant. Among them, *Quercus peduncularis* was the most important with an RVI of 48.35 at prescribed burning, 44.16 at non-burning, and 44.33 at wildfire sites. The second species of the higher order of importance was *Pinus oocarpa* for all burning sites, but its RVI decreased sharply in wildfire sites (Figure 3). At the same time, *Brysonima crassifolia*, a non-native species, increased in wildfire sites compared to non-burning and prescribed burning (Figure 3).

Table 2. Analysis of species composition similarity between sites with different burning types.

Sorensen's (Jaccard's) Similarity Indices between Burning Treatments			
	Prescribed Burning	Non-Burning	Wildfire
Prescribed burning		78 (64)	59 (42)
Non-burning			55 (38)
Wildfire			

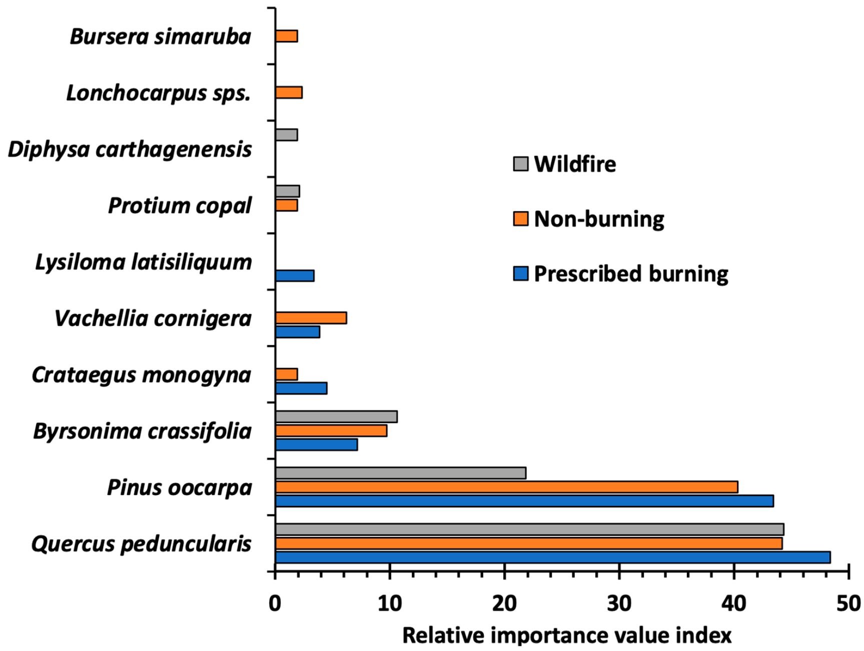

Figure 3. Relative importance value indices of tree species among three different fire types: prescribed burning, non-burning, and wildfire.

There was a higher similarity on species composition between prescribed burning and non-burning sites, but we observed the lowest similarity between wildfire and non-burning sites (Table 2).

3.2. Carbon Storage

The amount of carbon stored in live tree biomass varied from 32.67 to 52.04 Mg C ha^{-1} (Table 3). Despite the fact that wildfires reduced plant biodiversity, the live tree biomass did not differ significantly between burning types ($F = 1.61$, $p = 0.227$). However, the above- and belowground living biomass stocks tended to decrease in wildfire sites (Table 3). Forest floor litter stock tended to be higher in non-burning sites (4.65 Mg C ha^{-1}) but it did not differ statistically from prescribed burning (3.59 Mg C ha^{-1}) and wildfire sites (2.68 Mg C ha^{-1}).

Table 3. Carbon stocks in tree biomass, forest litter, deadwood materials, and SOC between prescribed burning, non-burning, and wildfire sites of tropical highland forests in Chiapas, Mexico. AGB = aboveground biomass, BGB = belowground (root) biomass.

Carbon Pools	Carbon Stocks (Mg ha^{-1}) Mean (95% Confidence Interval)		
	Prescribed Burning	Non-Burning	Wildfire
AGB (Mg C ha^{-1})	41.96 (28.85–55.06)	35.23 (22.12–48.33)	26.14 (13.03–39.24)
BGB (Mg C ha^{-1})	10.08 (7.10–13.05)	8.60 (5.62–11.58)	6.53 (3.56–9.51)
Total biomass (Mg C ha^{-1})	52.04 (35.96–68.12)	43.83 (27.75–59.91)	32.67 (16.59–48.75)
Deadwood (Mg C ha^{-1})	2.28 (1.06–3.51)	1.20 (0.00–2.43)	0.59 (0.00–1.82)
Litter (Mg C ha^{-1})	3.59 (2.33–4.86)	4.65 (3.38–5.92)	2.68 (1.42–3.95)
SOC (Mg C ha^{-1})	183.9 (166.1–201.7)	167.9 (150.1–185.7)	144.3 (126.5–162.1)

Mean SOC concentration ranged from 3.4% to 5.8%. SOC concentrations varied significantly between burning types ($F = 3.28$, $p = 0.039$) and soil depth classes ($F = 47.75$, $p < 0.001$), but the interaction between them was not statistically significant ($F = 1.39$, $p = 0.238$). In the upper 0–10 cm depth, prescribed burning showed a higher SOC concentration compared to wildfire, but it was statistically indifferent from non-burning sites (Figure 4). At lower depths, burning had no effects on SOC concentration (Figure 4).

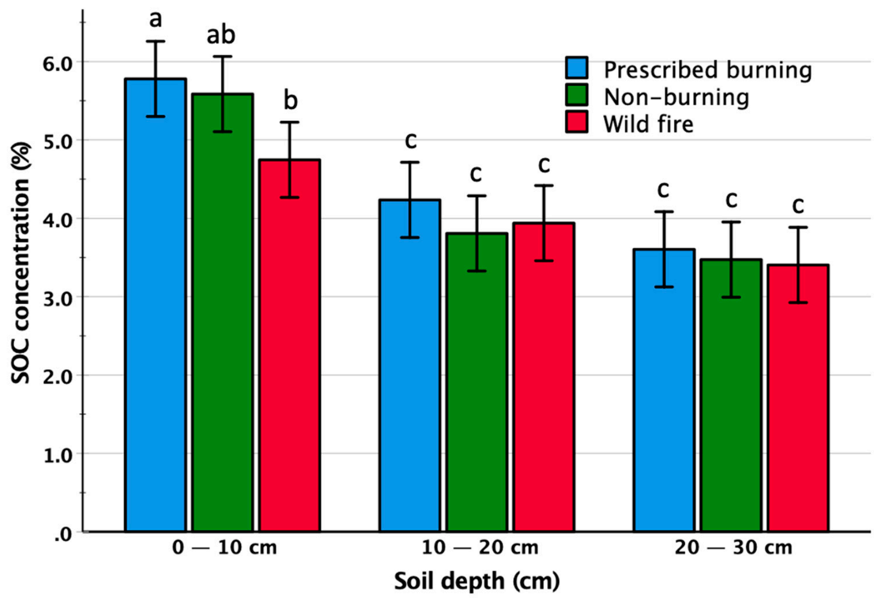

Figure 4. Soil organic carbon concentrations (%) by depth class among prescribed burning, non-burning, and wildfire sites. Different letters above the bars indicate statistically significant differences (Tukey $p < 0.05$) between burning treatments within the same depth class. Error bars correspond to the respective 95% confidence intervals.

SOC stock in the upper 0–10 cm soil at prescribed burning sites did not differ from non-burning sites but it was significantly higher than wildfire sites (Figure 5). SOC stock to 30 cm depth was significantly higher in prescribed burning sites (183.9 Mg C ha^{-1}) compared to wildfire sites (144.3 Mg C ha^{-1}). The difference between prescribed burning and non-burning sites (167.9 Mg C ha^{-1}) was non-significant ($p > 0.05$). The highest amount of carbon was stored in the upper 10 cm of soil compared to lower depth classes for prescribed burning and non-burning sites, but this difference between depth classes was not significant in wildfire sites (Figure 5).

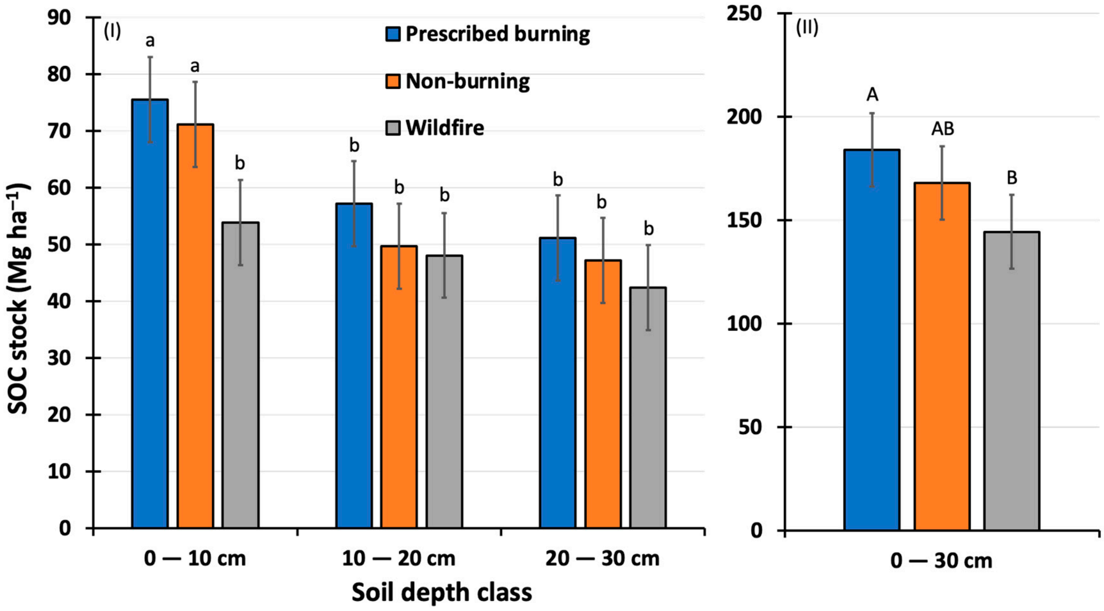

Figure 5. Soil organic carbon (SOC) stocks (Mg C ha^{-1}) among prescribed burning, non-burning, and wildfire sites at different depth classes (I), and the sum of SOC stocks down to 30 cm depth (II). Error bars denote 95% confidence intervals. Different letters above bars indicate significant differences (Tukey $p < 0.05$) between burning types within the same soil depth class.

3.3. Soil Properties

Burning treatment had a significant effect on some soil properties (Figure 6). The average soil bulk density varied between 1.2 and 1.4 g·cm^{-3}. When compared for each soil depth class, soil bulk density did not differ significantly between burning types. Soil pH values ranged from 4.4 to 4.7. Wildfire sites had significantly lower soil pH compared to prescribed burning and non-burning sites at all depth classes. Soil redox potential was higher at wildfire sites than at prescribed burned or non-burned sites. Prescribed burning and non-burning sites tended to have a greater soil moisture than wildfire sites (Figure 6), although the difference was not statistically significant. Fine root biomass to 30 cm depth tended to be higher in the prescribed burning sites compared to wildfires but did not differ statistically ($p > 0.05$).

3.4. Relationship between Variables of Carbon Storage, Soil, and Plant Diversity

Principal component analysis extracted four components with eigenvalues higher than 1. These components explained a cumulative 71% of the variance. The first component correlated more with soil pH, soil moisture, and tree diversity, while the second component correlated more with SOC content, tree biomass, bulk density, and fine root biomass. The third and fourth components correlated more with litter and deadwood mass, respectively (Table 4).

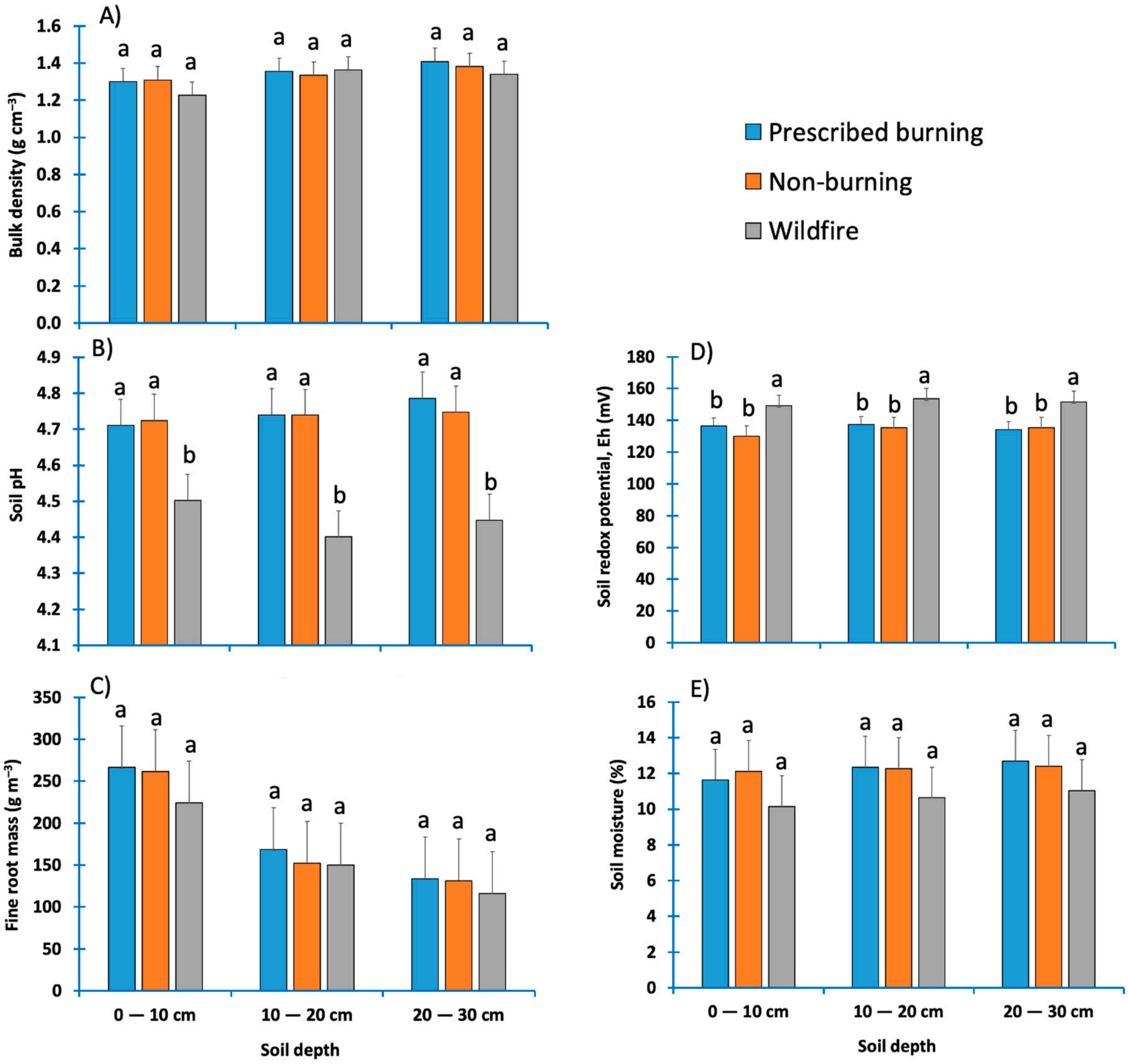

Figure 6. Soil properties and fine root biomass among prescribed burning, non-burning, and wildfire sites at different depth classes. (**A**) Soil bulk density, (**B**) soil pH, (**C**) fine root mass, (**D**) soil redox potential, and (**E**) soil moisture. Error bars indicate respective 95% confidence intervals. Different letters over the bars indicate significant differences between burning types ($p < 0.05$) within the same depth class.

Principal components 1 and 2 collectively explained 44% of the variance. In Figure 7, we plotted them, putting component 1 as the y-axis and component 2 as the x-axis. We observed that most of the prescribed burning and non-burning sites were distributed in the upper panel of the graph, indicating that these sites are characterized by higher tree diversity and soil pH. Wildfire sites are mostly distributed in the lower panel of the graph, indicating lower tree diversity and lower soil pH. In general, both non-burning and prescribed burning sites were distributed throughout the x-axis, but a greater number of sites appeared in the right panel, indicating the trend of higher carbon stocks. Wildfire sites are distributed in the left panel, showing lower carbon storage (Figure 7).

Table 4. Factor loadings extracted from principal component analysis using normalized varimax rotation. Correlations between principal components and predictor variables that are higher than 0.500 are marked in red.

Variables	Component 1 (Soil and Diversity)	Component 2 (Carbon Store)	Component 3 (Litter Accumulation)	Component 4 (Deadwood)
SOC content %	0.13	0.76	0.03	0.11
Soil pH	0.87	0.02	−0.16	0.06
Soil_moisture	0.66	−0.39	0.20	−0.39
Soil bulk density	−0.37	0.72	0.18	−0.22
Tree biomass	0.27	0.71	0.07	0.19
Litter mass	0.25	0.13	−0.77	−0.13
Deadwood mass	0.13	−0.12	0.01	−0.96
Fine root biomass	−0.15	0.56	−0.45	0.06
Tree diversity	0.86	0.21	−0.04	−0.18
Elevation	−0.41	−0.43	0.30	−0.03
Slope of the terrain	−0.04	−0.28	−0.84	0.16
Variance explained	22.15%	21.83%	15.47%	11.47%

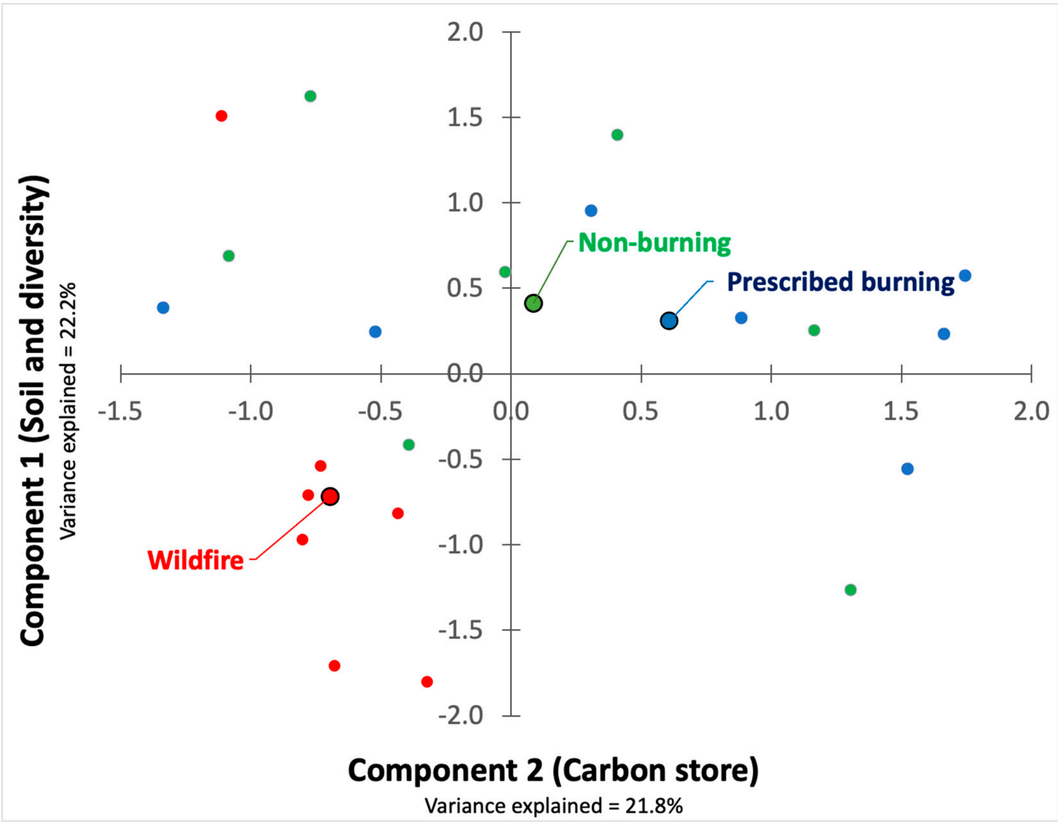

Figure 7. Biplot from principal component analysis taking into account the first two components that explained the major portion of the variance. Blue circles represent prescribed burning sites, green non-burning sites, red wildfire sites.

4. Discussions

Our results showed that wildfires have negative effects on forest biodiversity and carbon storage while prescribed burning showed a neutral effect on tree diversity and soil organic carbon storage if compared to non-burning sites. Although we did not observe a significant difference between burning treatments, prescribed burning did not lower the stocks of vegetation biomass compared to non-burning sites. There were some positive changes in soil properties as well, resulting from prescribed burning.

4.1. Burning Effect on Ecological Indices

Post-fire plant diversity depends on burning severity, plant adaptation to new conditions, availability of seed banks and canopy structure of the existing trees [48]. In our study, prescribed burning showed a higher plant diversity index compared to wildfire sites and did it not decrease the diversity compared to non-burning sites. Our findings are consistent with a study in the Brazilian Cerrado which reported that prescribed burning did not cause any loss of plant diversity [61]. Burning treatments often increase light availability for understory vegetation, decrease forest fuel load, and increase the availability of mineral nutrients [42]. Higher seed germination and lower sapling/tree mortality under prescribed burning can explain the greater plant diversity and species richness in these sites. Survival of trees after burning and recruitment of new trees have been found greater after a low-severity fire are important for higher tree diversity [62]. Controlled and planned application of fire in prescribed burning has a positive effect on litter layer thinning, which facilitates seeds of many plant species to reach the soil layer [63]. Low-intensity fire also helps to break the seed dormancy of many plant species favoring seed germination [34]. Furthermore, low-intensity fire modifies soil parameters that induce sapling establishment and provide essential mineral nutrients produced during litter burning. Distinct from high-intensity wildfires, the damage to saplings and adult trees is less severe in prescribed burning because fire managers constantly monitor the burning process to avoid severe damage [35]. We observed that the dominance index was higher at wildfire sites than at non-burning and prescribed burning sites. This is explained by the fact that only a few species can adapt and survive under the conditions of high-intensity wildfires [21].

We noted that the abundance of some non-native plant species increased at wildfire sites compared to non-burning o prescribed burning sites. Other studies have reported the effect of wildfires on the establishment and growth of exotic or invasive plant species. Forests in our study sites are dominated by *Quercus peduncularis*, *Pinus oocarpa*, and *Brysonima crassifolia*, but there was an increase in the relative importance value of *Byrsonima crassifolia* in wildfire sites. The high-intensity wildfire created open patches allowing more sunlight to arrive at the ground surface that might have induced germination of *B. crassifolia* and quick regeneration after a wildfire [64]. Some studies highlighted that prescribed burning should be carried out at low-intensity fires to promote understory vegetation growth and maintain diversity in ecosystems [19,32]. Burning-related changes in plant species diversity, stand structure, and primary productivity are important functional properties of ecosystems related to carbon storage [48,65].

4.2. Burning Effects on C Storage

Our results on the effects of burning treatment on stand structural properties are consistent with a study in the Blue Mountains of Oregon, USA, where prescribed burning did not affect tree basal area [66]. Minimal changes in stand structural properties and better ecological indices in prescribed burning sites explain the indifference of vegetation carbon storage in comparison to non-burning plots. Congruent to our findings, a study in subtropical forests of China showed that the tree biomass carbon stock difference between prescribed burning and non-burning sites was not significant [41].

Furthermore, we observed increased soil organic carbon stocks in prescribed burning sites compared to wildfire sites. Although prescribed burning reduced mineral-free soil organic carbon to 5 cm depth in the coniferous forests of California USA, it did not reduce mineral-associated organic carbon [24]. Low-intensity fire transformed organic matter to the pyrogenic recalcitrant form of carbon that is less sensitive to decomposition loss, and reduced heterotrophic soil CO_2 respiration by 55% [24], which could partly explain the higher SOC storage under prescribed burning in our study sites. A study in the pine forests of central Spain reported no marked changes in soil organic carbon content and quality due to prescribed burning [43]. Consistent with our results, a meta-analysis demonstrated that wildfire significantly reduced SOC storage compared to prescribed burning in Pacific Northwest forests of the USA [27]. Distinct from our findings, prescribed burning significantly reduced SOC storage and its recovery took more than one year of burning [67]. A modeling study calibrated in Australian mixed forests suggested that low-intensity prescribed burning at an interval of about 10 years did not harm the net ecosystem carbon balance (carbon sequestration rates) or the total ecosystem carbon storage; rather, it offsets the carbon loss from potential wildfire [68].

4.3. Burning Effects on Soil Properties

Soil physical, chemical, and biological property responses to prescribed burning vary between ecosystem types, fire frequency, and severity [28,69]. Both positive and negative changes in soil properties are reported [25]. In this study, small improvements were observed in soil pH, redox potential, and soil moisture content in prescribed burning sites in comparison to wildfire sites. Soil properties were similar to non-burning sites in this study. A study in *Pinus resinosa* forest in northern Minnesota showed that prescribed burning has desirable effects on soil properties but the effects depended on the season of the burning [70]. In a sagebrush steppe ecosystem of the Columbia Basin, USA, wildfire significantly reduced soil organic carbon and enzymatic activity relative to non-burning sites [71]. Changes in soil chemical properties due to the single prescribed burning treatment were negligible compared to unburned plots in Fishburn Forest, Virginia, USA [72]. Low-intensity prescribed burning may change soil properties and reduce the soil organic matter decomposition rates, leading to higher accumulation of soil carbon compared to wildfires [73].

5. Conclusions

The results showed that prescribed burning conserved plant biodiversity at the level of non-burning and more so than wildfire. Species composition showed a greater similarity between sites with prescribed burning and without fire disturbance compared to the sites with wildfires. The effect of prescribed burning on vegetation biomass stocks was not significant compared to unburned and wildfire sites. However, prescribed burning had a significant effect on soil organic carbon stocks, with a greater amount of carbon stored compared to wildfire sites. Soil organic carbon stocks in prescribed burning sites were statistically indifferent when compared to non-burning sites. The multivariate analysis demonstrated that variables such as soil organic carbon content, fine root biomass, and living tree biomass correlated positively, indicating that higher aboveground biomass has a positive effect on belowground carbon storage in these forest ecosystems. Furthermore, better soil properties in prescribed burning and non-burning sites were linked to higher plant diversity. Our results demonstrated that prescribed burning is useful, as it maintained plant diversity and carbon stocks to the level of non-burned areas in fire-dependent ecosystems. Wildfire sites showed a decrease in biodiversity indices and carbon storage. For future studies, it is recommended to take into account the effect of prescribed burning on the availability of different nutrients and soil biological diversity that are of great importance in ecosystem functioning. These results could help in making decisions on fire management to prevent severe damage from wildfires that can emit large amounts of carbon stored in forest vegetation and soil.

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