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Monitoring of a Seedling Planting Restoration in a Permanent Preservation Area of the Southeast Atlantic Forest Biome, Brazil

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Abstract: The Atlantic Forest biome is a biodiversity hotspot with only 13% of its native vegetation cover remaining in Brazil. Forest restoration is an important process for the re-establishment of biodiversity and the preservation of water resources in degraded areas, like riparian forests. Monitoring is an essential step of restoration, because the periodic evaluation of indicators allows researchers to analyze the effectiveness of applied techniques. This study aimed to evaluate the environmental quality of a seedling planting (active restoration) in a Permanent Preservation Area (PPA) of the Southeast Atlantic Forest biome, through a monitoring protocol, within a year. More specifically, we aimed to investigate the following questions: (1) do some ecological groups or families grow more than others? and (2) is the cover of exotic grasses negatively influencing forest regeneration? Data were collected during November 2017 and April (interval represents five months of wet season) and October 2018 (six months of dry season). Eight plots of 9 × 18 m were established and all individuals were identified and measured (diameter at ground level (DGL), height (H), and canopy diameter (CD)). Exotic grass cover and richness of regenerating species were also recorded. We registered 119 individuals during the three inventories, distributed in 35 species and 14 families. Results indicate an inverse association between exotic grasses and the presence of recruits in the area. Thus, exotic grass control may be needed while the forest canopy is not closed. A significant growth of individuals in the whole study period, especially during the wet season, was detected. Legumes grew more in trunk and canopy diameter (*Anadenanthera colubrina* (Vell.) Brenan and *Inga vera* Willd. subsp. *affinis* (DC.) T. D. Penn. had the biggest growth, respectively) than non-legumes during wetter months and within a year. Pioneers had greater height increase compared to non-pioneers only during wetter months. Legumes may be important species to be used in other young tropical forest restoration areas.

Keywords: atlantic forest; restoration plantation; native species; woody species

1. Introduction

The Atlantic Forest biome is a biodiversity hotspot [1], and only 13% of its native vegetation cover remains in Brazil [2] due to intense deforestation and human disturbance that occurred mostly in the first half of the 19th century [3]. Nature reserves protect only 9% of the remaining forest [4]. Recent (last five decades) demographic and market shifts have resulted in rural land abandonment, which in turn has transitioned some portions of the biome back to forest [5]. In Southeast Atlantic Forest sites, the forest transition tends to occur in degraded pasture areas that are less suitable for

agriculture. The Atlantic Forest biome has become an important global conservation and restoration focus [6]. Intense efforts by non-governmental organizations to recover native Brazilian ecosystems are occurring [7] and, locally, many isolated restoration initiatives are taking place.

In the last decade, forest restoration has grown rapidly in Brazil, mainly due to the increasing demand for legal environmental regularization of agricultural properties (due to the Rural Environmental Registry [8,9]); the mitigation and compensation of environmental impacts; and the intense efforts of governments and non-governmental organizations in the recovery of native ecosystems through, for example, the National Policy for the Recovery of Native Vegetation of 2017 [10], which articulates, integrates, and promotes actions for the recovery of forests and other forms of native vegetation. One of these efforts, the Atlantic Forest Restoration Pact in Brazil, is a national coalition with the goal of promoting the restoration of 15 million ha of the Atlantic Forest by 2050 [11,12]. Some of the main initiatives of Atlantic Forest restoration are carried out in Permanent Preservation Areas (PPAs), like riparian forests, with the environmental function of preserving water resources [13].

Active and passive restoration are two important strategies to aid the recovery of large areas of deforested and degraded tropical lands [14], and they respond to the legal needs of PPA recomposition (see article 19 in [10]). In general, the presence of nearby old-growth forest remnants and high levels of seed dispersal make recovery (passive restoration) possible [15]; however, under high disturbance conditions and with an absence of forest remnants a site may never return to a state similar to the original, and active restoration is needed [16]. The planting of tree seedlings is one of the most used techniques for active forest restoration in Atlantic forests and in the studied region [17–19]. This practice commonly uses two functional groups—filling and diversity. The first one corresponds to species that grow fast and have good canopy cover, causing a quick closing of the restored area. On the other hand, the second one refers to species that do not have rapid growth or good cover, but are essential for the perpetuation of a forest, including species that occupy distinct successional stages (pioneers, secondary, and climax) [20].

There are few works analyzing the restoration of tropical (Atlantic, in this case) forests in the world. These studies mainly show that past restoration projects have not produced high diversity self-perpetuating forests in the Brazilian Atlantic Forest [21,22], and that planting trees alone is insufficient to drive the rapid recovery of the rain forest plant community [23,24]. Despite that, planting seedlings is expected to accelerate and increase the predictability of a degraded site to become a rain forest physiognomy [25], because they reconcile high species richness and diversity with good wood volume yields and tree biomass [26], being then preferred over cheaper restoration approaches as passive restoration. Additionally, older (five years) actively restored areas in the neotropical rain forest had a lower probability of being covered by invasive grasses and a higher probability (57%) of having a closed canopy than passive restoration sites [19].

The assessment and monitoring of restored forests are essential for correcting and improving restoration techniques, especially in tropical ecosystems [27]. Monitoring is possibly the most important step in a restoration project, since it allows researchers to evaluate, through indicators, the responses of a degraded area to treatments. Some studies made in a tropical environment highlight the importance of monitoring to restoration [22,23]. However, the difficulty in carrying out these studies relates to the fact that monitoring is performed mostly to fulfill legal commitments [27]. In addition, the lack of consensus in scientific literature on the most appropriate indicators for assessing the success of tropical forest restoration and, consequently, the associated environmental gains and high costs of implementation is an important obstacle for monitoring neotropical rain forest restoration [27]. Most studies argue that these indicators are influenced by restoration ages [19,22,28], and that studies should focus on the composition, structure, and functioning [27] and defend the establishment of a monitoring protocol for all tropical regions [12,29–31].

Restoration plantings carried out in the Atlantic Forest of southeastern Brazil have shown predictable trajectories in terms of vegetation structure and species richness [32], but they were based in old restoration sites that had already enough canopy cover to support succession. Many younger

restoration projects may not reach this stage, and be lost before the canopy is close enough to shade invasive grasses and support the recolonization of woody native species in the understory [19], but studies about these young sites are scarce. It is highly important early on to evaluate restored area success with respect to the invasion of exotic grasses and the increase of canopy cover. In the Atlantic Forest, it is widely known that native species richness increases while excluding invasive grasses [33]. Thus, this study aimed to evaluate the environmental quality of a seedling planting (active restoration) in a permanent preservation area (watercourse marginal strip) of the Southeast Atlantic Forest biome, through a monitoring protocol [12], within a year. More specifically, we aimed to investigate the following questions: (1) do some ecological groups or families grow more than others? and (2) is the cover of exotic grasses negatively influencing forest regeneration? We expected the following: (i) pioneers and legumes would grow more than other species during the study period; (ii) exotic grass cover would be low; and (iii) the number of regenerating species (recruits) would be high and increase with time in the site. Despite the short time between inventories, this study is important to better understand restoration in the Atlantic Forest, especially in the early stage.

2. Materials and Methods

2.1. Study Area

This study was conducted in a 1 ha restored permanent preservation area (PPA) within a cattle ranch (12.45 ha), located between the coordinates 23°5'59.02'' S and 45°57'29.07'' W, in São José dos Campos, Southeast Atlantic Forest biome, São Paulo state, Brazil (Figure 1). The region has a strong hilly relief, with a 20% to 45% slope [34], and the soil is red-yellow latosol [35]. The climate is classified as dry-winter subtropical (Cwa), according to Köppen [36], with an annual mean temperature of 20.9 °C, an average annual precipitation of 1088.5 mm, and a dry season between April and September (data collected at the Meteorological Station of Centro de Treinamento da Aeronáutica in São José dos Campos). The ranch is in the Atlantic Forest biome (transition between evergreen and deciduous forest).

Figure 1. Satellite image of the study site, inside São José dos Campos municipality, São Paulo state, Brazil.

The 1 ha area was a low-intensity cattle pasture until 2014, when it was fenced. Exotic grass was controlled and seedlings of woody species were planted in 2015, by a non-governmental organization called Associação Corredor Ecológico Vale do Paraíba. Planted seedling species were mostly from native species of the Atlantic Forest biome. They were planted in a 3×2 m spacing design and only in areas without old trees and shrubs. Total planted seedlings were about 1000 per hectare.

2.2. Plant Measurements

In November 2017 we randomly established eight 9×18 m plots, according to the monitoring protocol of the Mata Atlântica Restoration Pact [20]. All planted individuals (>3 cm of circumference at ground level) inside the plots were tagged, identified, and measured. Plant material was identified using botanical identification keys [37,38]. For plant family classification, we used the Angiosperm Phylogeny Group IV [39] and the Brazil Flora List [40]. Some seedlings had only a few or damaged leaves that made the identification difficult. Both the non-governmental organization and the ranch owner lost track of the species initially planted.

Inventories were carried out in November 2017 (survey 1 (S1), wet season), April 2018 (S2, dry season), and October 2018 (S3, wet season) to understand growth differences between seasons. Additionally, forest restoration projects of 1 to 3 years (post-implantation), like the studied site, require biannual monitoring because they may still need corrections [20]. Measurements of seedlings and young trees were the circumference at ground level (transformed into diameter at ground level (DGL) as circumference/ π), the plant height (H), and the canopy diameter (CD). Each plot had four 1×1 m subplots, one in each plot vertex, where regenerating seedling species (identified by morphospecies) and the percentual cover of exotic grasses were quantified in November 2017 and April 2018. In October 2018, exotic grasses were suppressed with herbicides, making the third survey impossible. Species were classified into two family groups (legumes and non-legumes) and according to the ecological group into pioneers and non-pioneers classified using the studies [41,42], the dispersal syndrome (zoo, auto, and anemochory) using [41,42], and the leaf deciduousness (deciduous, semi-deciduous, and evergreen) using [43]. All functional groups were mutually exclusive.

2.3. Statistical Analysis

All individuals were registered in the three inventories; thus, we compared H, DGL, and CD at individual levels between inventories (S1, S2, and S3) and we used nonparametric tests, because data distribution was not normal (Shapiro–Wilk tests, $p < 0.05$ for all variables). Nonparametric estimation is a safe strategy in restoration sites, with multiple young individuals and left skewed data distribution. We used the Wilcoxon signed-rank test for paired samples of H, DGL, and CD to observe variations in plant measurements between inventories. We also compared H, DGL, and CD by family and ecological group using the Wilcoxon signed-rank test. Additionally, the Pearson correlation test was used to evaluate the association between the number of regenerating species and the percentual cover of exotic grasses. Statistical analyses were performed in R software version 3.5.0 (R Core Team, R Foundation for Statistical Computing, Vienna, Austria) [44]. Data are shown as median (with first and third quartiles).

3. Results

We registered 119 individuals during the three inventories, distributed in 35 species and 14 families (Table 1). The most abundant families were Fabaceae and Myrtaceae (each with nine and five species, respectively), followed by Bignoniaceae and Anacardiaceae (with four and two species, respectively, see Table 1). *Inga vera* Willd. subsp. *affinis* (DC.) T. D. Penn. (15) and *Cecropia hololeuca* Miq. (13) were the most abundant species (Table 1). Plots had an average of 15 individuals each and the total density was 926 individuals/ha.

Pioneers were the most abundant ecological group (66 individuals), but not the most frequent (11 species, see Table 1). Non-pioneers had 14 species and 34 individuals (Table 1). Zoochory predominated (18 species, 72 individuals), followed by autochory (6 species, 27 individuals) and anemochory

(6 species, 12 individuals, see Table 1). Deciduous species were most frequent and abundant (9 species, 38 individuals), followed by semi-deciduous (8 species, 34 individuals) and evergreen (8 species, 28 individuals, see Table 1). *Anadenanthera colubrina* (Vell.) Brenan, *Inga laurina* (Sw.) Willd., *Inga marginata* Willd., and *Poincianella pluviosa* (DC.) L.P. Queiroz were Fabaceae and non-pioneers (Table 1).

Table 1. List of the studied species, containing the number of sampled individuals (N); the ecological group (EG)—pioneers (P) and non-pioneers (NP); the dispersal syndrome (SD)—anemochoric (ANE), autochoric (AUTO), and zoochoric (ZOO); and the leaf deciduousness (LD)—deciduous (D), evergreen (E), and semi-deciduous (S).

Family	Species	N	EG	SD	LD
Anacardiaceae	<i>Schinus molle</i> L.	1	P	ZOO	E
	<i>Schinus terebinthifolia</i> Raddi	4	P	ZOO	E
Bignoniaceae	<i>Handroanthus chrysotrichus</i> (Mart. ex D.C.) Mattos	3	NP	ANE	D
	<i>Handroanthus impetiginosus</i> (Mart. ex D.C.) Mattos	2	NP	ANE	D
	<i>Handroanthus</i> sp.	1	-	ANE	-
Connaraceae	<i>Handroanthus umbellatus</i> (Sond.) Mattos	1	NP	ANE	D
Euphorbiaceae	<i>Connarus regnellii</i> G. Schellenb.	2	NP	ZOO	E
Fabaceae	<i>Croton</i> sp.	1	-	AUTO	-
	<i>Anadenanthera colubrina</i> (Vell.) Brenan	8	NP	AUTO	D
	<i>Dalbergia nigra</i> (Vell.) Allemão ex Benth.	2	P	AUTO	D
	<i>Inga laurina</i> (Sw.) Willd.	1	NP	ZOO	E
	<i>Inga marginata</i> Willd.	1	NP	ZOO	S
	<i>Inga</i> sp.	7	-	ZOO	-
	<i>Inga vera</i> Willd. subsp. <i>affinis</i> (DC.) T. D. Penn.	15	P	ZOO	S
	<i>Piptadenia gonoacantha</i> (Mart.) J.F. Macbr.	5	P	AUTO	S
	<i>Poincianella pluviosa</i> (DC.) L.P. Queiroz	1	NP	AUTO	S
	<i>Schizolobium parahyba</i> (Vell.) Blake	10	P	AUTO	D
Lauraceae	<i>Ocotea</i> sp.	1	-	ZOO	-
Malpighiaceae	<i>Byrsonima crassifolia</i> (L.) Kunth	1	P	ZOO	S
Malvaceae	<i>Ceiba speciosa</i> (A.St.-Hil.) Ravenna	1	NP	ANE	D
Melastomataceae	<i>Tibouchina granulosa</i> (Desr.) Cogn.	4	P	ANE	S
	<i>Campomanesia guazumifolia</i> (Cambess.) O. Berg	1	P	ZOO	D
Myrtaceae	<i>Eugenia stipitata</i> Mc Vaugh	1	NP	ZOO	E
	<i>Plinia edulis</i> (Vell.) Sobral	4	NP	ZOO	E
	<i>Psidium guajava</i> L.	6	NP	ZOO	S
	<i>Psidium guineense</i> Sw.	2	NP	ZOO	E
Primulaceae	<i>Myrsine</i> sp.	1	-	ZOO	-
Sapindaceae	<i>Cupania vernalis</i> Cambess.	1	NP	ZOO	S
Urticaceae	<i>Cecropia hololeuca</i> Miq.	13	P	ZOO	E
Verbenaceae	<i>Citharexylum myrianthum</i> Cham.	10	P	ZOO	D
Unknown	Unknown 1	1	-	-	-
	Unknown 2	1	-	-	-
	Unknown 3	1	-	-	-
	Unknown 4	1	-	-	-
	Unknown 5	4	-	-	-
		119			

Between S1 and S2, *Anadenanthera colubrina* (Vell.) Brenan, Unknown 4, and *Connarus regnellii* G. Schellenb grew more in DGL, H, and CD, respectively. Between S2 and S3, *Dalbergia nigra* (Vell.) Allemão ex Benth. grew more in DGL and CD and *Cupania vernalis* Cambess. had higher H growth. Between S1 and S3, *Anadenanthera colubrina* (Vell.) Brenan, Unknown 4, and *Inga vera* Willd. subsp. *affinis* (DC.) T. D. Penn. grew more in DGL, H, and CD, respectively.

All 119 individuals had significant growth between S1 and S2 for DGL (1.3 cm; $W = 71.5$; $p < 0.001$), H (0.5 m; $W = 75.5$; $p < 0.001$), and CD (0.3 m; $W = 749.5$; $p < 0.001$). Between S2 and S3, all 119 individuals grew significantly for DGL (0.3 cm; $W = 1012.5$; $p < 0.001$) and H (0.1 m; $W = 1259.0$;

$p < 0.001$), but not for CD (0 m; $W = 2831.0$; $p = 0.989$). Comparing the whole experiment period, all individuals had significant growth between S1 and S3 for DGL (1.75 cm; $W = 118.0$; $p < 0.001$), H (0.7 m; $W = 200.5$; $p < 0.001$), and CD (0.4 m; $W = 1169.0$; $p < 0.001$).

Legumes grew more in DGL, H, and CD than non-legumes, between S1 and S2, and in DGL and CD between S1 and S3 ($n = 111$; Table 2). There were no differences in DGL, H, and CD by family between S2 and S3 (Table 2). Pioneers did not differ from non-pioneers in DGL, H, and CD for all periods; the exception was a higher height increment of pioneers between S1 and S2 than of non-pioneers ($n = 100$; Table 3).

Table 2. Growth in diameter at ground level (DGL), plant height (H), and canopy diameter (CD) of all individuals ($n = 111$) by family (legumes and non-legumes), among the three surveys (S1, November 2017; S2, April 2018; and S3, October 2018). Significant differences ($p < 0.05$) are in bold. Data are shown as median (with first and third quartiles).

Interval	Parameters	Legumes	Non-Legumes	<i>p</i> -Value	Test
S1–S2	DGL (cm)	1.6 (0.8; 2.5)	1.1 (0.5; 1.6)	0.005	$W = 1780.5$
	H (m)	0.6 (0.4; 1.0)	0.4 (0.2; 0.8)	0.024	$W = 1695.5$
	CD (m)	0.6 (0.1; 0.9)	0.2 (0; 0.5)	0.028	$W = 1685.5$
S2–S3	DGL (cm)	0.4 (0; 1.0)	0.3 (0; 0.6)	0.421	$W = 1416.0$
	H (m)	0.1 (0; 0.4)	0.1 (0; 0.4)	0.321	$W = 1147.5$
	CD (m)	0.1 (0; 0.7)	0 (0; 0.1)	0.122	$W = 1527.0$
S1–S3	DGL (cm)	2.3 (1.4; 3.3)	1.6 (0.6; 2.3)	0.002	$W = 1972.5$
	H (m)	0.8 (0.4; 1.1)	0.6 (0.3; 1)	0.178	$W = 1697.0$
	CD (m)	0.7 (0.4; 1.2)	0.3 (0; 0.5)	<0.001	$W = 2114.0$

Table 3. Growth in diameter at ground level (DGL), plant height (H), and canopy diameter (CD) of all individuals ($n = 100$) by ecological groups (pioneers and non-pioneers), among the three surveys (S1, November 2017; S2, April 2018; and S3, October 2018). Significant differences ($p < 0.05$) are in bold. Data are shown as median (with first and third quartiles).

Interval	Parameters	Pioneers	Non-Pioneers	<i>p</i> -Value	Test
S1–S2	DGL (cm)	1.4 (0.8; 2.2)	1.2 (0.6; 1.8)	0.206	$W = 846.5$
	H (m)	0.7 (0.3; 1)	0.5 (0.2; 0.7)	0.015	$W = 698.0$
	CD (m)	0.4 (0; 0.7)	0.2 (0; 0.9)	0.843	$W = 981.0$
S2–S3	DGL (cm)	0.5 (0; 0.9)	0.3 (0; 0.6)	0.481	$W = 887.0$
	H (m)	0.2 (0; 0.4)	0.1 (0; 0.4)	0.295	$W = 1102.5$
	CD (m)	0 (0; 0.5)	0 (0; 0.2)	0.435	$W = 877.0$
S1–S3	DGL (cm)	1.9 (1.3; 2.9)	1.6 (0.6; 2.7)	0.107	$W = 871.5$
	H (m)	0.8 (0.4; 1.2)	0.6 (0.4; 1)	0.433	$W = 982.5$
	CD (m)	0.5 (0.1; 0.9)	0.4 (0; 0.6)	0.238	$W = 929.5$

Grass cover increased between S1 and S2 (from 25% to 80%; $W = 113.0$; $p < 0.001$), but the number of regenerating morphospecies did not (from 3 to 3; $W = 601.0$; $p = 0.226$). On S1, grass cover did not affect the number of regenerating morphospecies ($\rho = 0.146$; $p = 0.424$; Figure 2A); however, on S2, there was an inverse association between grass cover and regenerating morphospecies richness ($\rho = -0.443$; $p = 0.011$; Figure 2B).

Figure 2. Number of regenerating morphospecies related to the cover of exotic grasses in (A) November 2017 and (B) April 2018.

4. Discussion

Our results showed significant growth of individuals in the whole study period, especially during the wet season, that legumes grew more in diameter and canopy than non-legumes, and that pioneers had greater height increase compared to non-pioneers. These results indicate precipitation as an abiotic factor promoting growing of young seedlings in restoration sites of the Atlantic Forest. They also point to the conclusion that pioneers grow taller, but legumes spread laterally and may close the forest floor. These two different plant strategies may be used solely or in combination to improve the occupation in restored areas. Additionally, an inverse relationship between exotic grasses and the presence of recruits in the area was detected, proving the importance of monitoring and taking actions to control invasive species in the ecological restoration of neotropics.

We found Fabaceae and Myrtaceae as the most abundant families in the restored area. Fabaceae and Myrtaceae have been registered as two of the most important woody families in tropical forests in Brazil (evergreen in [45,46] and deciduous forest in [47,48]), hence their great use in forest restoration projects. It is also well known that Fabaceae plants grow faster than other families and produce great amounts of aboveground organic material, promoting soil chemical, physical and biological recuperation, particularly if used as green manure crops when the whole plant is plowed down into the soil [49,50]. Moreover, some legumes are capable of biological N_2 fixation [51]. Additionally, fleshy fruits of Myrtaceae plants feed fauna, promoting ecological interactions and seed dispersal in the restored area [52,53] and may encourage the economical use of native fruit, ornamentation, and medicinal species by the local farmers in restoration projects.

Pioneer, zoochorous, and deciduous species were the most abundant species in the restored area. Vegetation dynamics processes in tropical forests include colonization by shade-intolerant and shade-tolerant pioneer trees as one of the first phases of succession [54]. Thus, a high abundance of pioneers contributes to the initial recovering of restoring areas [20]. However, a few abundant pioneer species do not sustain restoration in the long term [27]. Zoochory predominates in tropical rainforests [55] and the seed–disperser interaction plays an important role in the natural regeneration dynamics of secondary and restoring forests [14]. Leaf deciduousness is more frequent in dry and deciduous forests and savannas in the tropics [56], but the restored area region is in a transition

zone between evergreen and semi-deciduous forests. Leaf deciduousness brings seasonal clearings in the vegetation and consequently light level changes to the forest ground, which may favor seed germination and colonization of pioneer species [57]. On the other hand, the increase in light level and temperature inside a forest environment may cause seed drying and death, decreased seedling establishment, invasion by exotic grasses, and ultimately damage to the restoration project progress [27,58].

Plants grew during all study intervals, but growth was higher between S1 and S2 (five months of wet season) than between S2 and S3 (six months of dry season). Other studies with Atlantic Forest species have shown greater tree growing during the rainy season than during the drought period [59,60]. Water deficit reduces plant metabolism and growing [61]. The significant plant growing during the one-year study period is attributed to the restored area age (three years old); for example, individuals grow more in the beginning of a forest development and generally growth rate decreases with tree age [62]. Fast-growing tree species, as well as legumes of this study, contributed significantly to the carbon stock during the early years (approximately 37 years) of restoration in the Brazilian Atlantic Forest [63]. In addition, canopy cover, basal area, and tree height significantly increased in other restoration areas (site between 12 and 55 years) in the Atlantic Forest [24].

In general, legumes had higher diameter and canopy growing than non-legumes (in the dry season, differences were not significant and height was also similar along the whole study, between legumes and non-legumes). *Anadenanthera colubrina* (Vell.) Brenan, *Dalbergia nigra* (Vell.) Allemão ex Benth., and *Inga vera* Willd. subsp. *affinis* (DC.) T. D. Penn. were the species that grew more during the study time. Fabaceae plants can grow faster than others [47,48]. This fast growth is influenced by *Rhizobia* (bacteria that fix nitrogen after becoming established inside root nodules), which produces biological N₂ fixation and stimulates root hair spread, where most water and nutrient absorption occurs [64]. When pioneers were compared to non-pioneer species (secondary and climax), most of the three evaluated size-related parameters were not different between them, indicating that the use of legumes in the beginning of a forest restoration project may be more recommended to promote a forest environment. Organizing species from different successional groups into functional groups for purposes of restoration has been previously suggested [20]. In fact, the use of Fabaceae species, as green manure or not, is increasing in tropical forests. Campoe et al. (2010) have shown that a higher proportion of pioneer species did not significantly alter the rate of biomass accumulation throughout a restoration site in the Atlantic Forest, but the inclusion of nitrogen-fixing species (Leguminosae) did [65].

There was a significant increase in exotic grass cover between S1 and S2 (five months of wet season), probably due to exotic grass phenology, which is usually influenced by rainfall [66]. Exotic invasion is one of the main reasons for the failure of tropical forest restoration [67], therefore, weed control is being recommended to reduce environmental stress, resulting in growth and survival of rainforest planted trees [65] and increasing the carbon stock [68]. Indeed, we noticed a negative effect of exotic grasses (especially *Brachiaria decumbens*) in regeneration, indicated by the richness of morphospecies. This result was also demonstrated in other studies [66,69]. Despite the grass cover influence, regeneration did not change in the same period of time. The landscape of the studied restored area comprises small patches of forest and great pasture areas; thus, long-distance seed dispersal may not be effectively happening. However, even if seeds of different species did arrive in the studied plots, the increase in exotic grass cover between S1 and S2 would suffocate and inhibit regeneration, which is indicated by the unchanged morphospecies richness.

5. Conclusions

In a restored permanent preservation area of the Southeast Atlantic Forest biome, we found the following: (i) legumes had a higher growth in trunk and canopy diameter than non-legumes within a year, so the former may be important species to be used in other young tropical forest restoration areas and (ii) high exotic grass cover is preventing natural regeneration in the site, indicated by the richness of morphospecies. The monitoring of restored forests is essential for correcting and improving restoration

techniques, especially in tropical ecosystems. Thus, exotic grass control may be needed while the forest canopy is not closed. It is also important to keep monitoring the area for a longer period, because the restoration of a PPA in a hilly relief with a steep slope is very sensitive to water erosion. As has been recently found, fostering the establishment and maintenance of seedling planting restoration can be an effective strategy for biodiversity recovery and biomass accumulation in restored forests [70].

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