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Influence of Removal of a Non-native Tree Species *Mimosa caesalpiniifolia* Benth. on the Regenerating Plant Communities in a Tropical Semideciduous Forest Under Restoration in Brazil

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Abstract Exotic species are used to trigger facilitation in restoration plantings, but this positive effect may not be permanent and these species may have negative effects later on. Since such species can provide a marketable product (firewood), their harvest may represent an advantageous strategy to achieve both ecological and economic benefits. In this study, we looked at the effect of removal of a non-native tree species (*Mimosa caesalpiniifolia*) on the understory of a semideciduous forest undergoing restoration. We assessed two 14-year-old plantation systems (modified "taungya" agroforestry system; and mixed plantation using commercial timber and firewood tree species) established at two sites with contrasting soil properties in São Paulo state, Brazil. The experimental design included randomized blocks with split plots. The natural regeneration of woody species (height $\geq$0.2 m) was compared between managed (all *M. caesalpiniifolia* trees removed) and unmanaged plots during the first year after the intervention. The removal of *M. caesalpiniifolia* increased species diversity but decreased stand basal area. Nevertheless, the basal area loss was recovered after 1 year. The management treatment affected tree species regeneration differently between species groups. The results of this study suggest that removal of *M. caesalpiniifolia* benefited the understory and possibly accelerated the succession process. Further monitoring studies are needed to evaluate the longer term effects on stand structure and composition. The lack of negative effects of tree removal on the natural regeneration indicates that such interventions can be recommended, especially considering the expectations of economic revenues from tree harvesting in restoration plantings.

Keywords Adaptive management · Atlantic forest · Seasonal semideciduous forest · Ecological restoration · Natural regeneration · Thinning

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Introduction

Ecological intensification strategies such as the use of nitrogen-fixing trees in mixed plantations have been demonstrated to increase both soil fertility and biomass accumulation (Voigtlaender et al. 2011; Siddique et al. 2008). Many non-native species have attributes such as fast growth, potential for improvement of soil physical and chemical properties, among other characteristics, which favor their use in restoration projects (D'Antonio and Meyerson 2002; ITTO 2005), provided their role in the ecosystem is clearly defined. However, after canopy development and weed infestation suppression, their ecological role as facilitators may decrease over time. Furthermore, these species may compete with other species, altering the dominance relations and possibly affecting the successional trajectory of the restored forest community. More studies are required to better understand the long-term effects of non-native species in restored native communities.

Interactions such as competition and facilitation have long been recognized as driving forces in plant community assembly (Callaway and Walker 1997). Some plant species can have strong direct and indirect effects on other plant species, and alterations in the community structure can have strong effects on the community organization as a whole. In particular, species removal can result in unexpected effects in the managed ecosystem (Zavaleta et al. 2001; Norton 2009). As in natural processes, forest management interventions such as thinning or timber harvesting generate gaps that change forest structure and may trigger natural regeneration (Lamprecht 1990) by influencing the germination and development of forest species in different ways (e.g. Denslow 1987; Alvarez-Buylla and Martinez-Ramos 1992; Dalling and Hubbell 2002). These changes in physical and biotic conditions, as well as the disturbance itself can be viewed as filters that may allow or prevent the entrance of new species in the community (Temperton and Hobbs 2004).

The use of pioneer non-native species in restoration planting designs can be justified by their possible facilitation effects (sensu Connell and Slatyer 1977; Padilla and Pugnaire 2006), particularly when these species have some economic value, thereby reconciling economic, social, and ecological aspects of the restoration (Lamb 1998, Lamb et al. 2005; Ewel and Putz 2004). This is important since ecological restoration is often very costly, and therefore a barrier to adoption by small landholders. Thus, the development of quick, efficient (e.g. Corbin and Holl 2012), and economically viable (Brancalion et al. 2012) strategies to restore ecosystems which integrate social and ecological benefits (Cao et al. 2014) can enhance the chances of restoration success (Brown et al. 2011).

The beneficial effect of the removal of different tree species on the regeneration of native species has been reported previously in different regions of the world. Ashton et al. (1998), studying different thinning intensities of an 18-year-old *Pinus* plantation in southwest Sri Lanka, observed generally higher growth and lower mortality rates under canopy removal treatments. Kueffer et al. (2010), testing the effects of removal of an invasive species (*Cinnamomum verum*) from native tropical secondary forest on the island of Mahé, Republic of Seychelles, concluded that there was a benefit for regeneration of native species over invasive species when the removal of the invasive species resulted in small canopy openings. Belote et al. (2012) in a study on the structure and composition of communities of oak forests in the Appalachian Mountains in Virginia, USA, after disturbance by logging, recorded an increased diversity after tree harvests. The same authors stated that the proportion of non-native plant species tended to increase following large scale disturbances, but tended to decrease over time due to canopy closure, suggesting that the provision of resources by disturbance had a strong influence on the species colonization and community composition. However, studies aiming at identifying patterns of response to this type of management in ecosystems undergoing restoration are scarce, despite its importance.

We studied the effects of eliminating a non-native tree species (*Mimosa caesalpiniifolia* Benth.) on the regenerating plant communities in tropical semideciduous forests undergoing restoration. *Mimosa caesalpiniifolia* (locally known as "sansão-do-campo" or "sabiá") has a variable growth habit, ranging from a shrub to an evergreen tree. The largest trees can reach up to 10 m in height and 30 cm in DBH (diameter at breast height, 1.30 m above soil). It produces a broad, sparsely branched canopy and multiple stems that are usually armed with prickles. Its natural range extends from 2°30′S to 15°20′S, at elevations between up to 400 m. It is a pioneer species that occurs in both primary and secondary formations, where it is common or frequent in old fields. It occurs in both Caatinga (dry forests) and Cerrado ecosystems (from savanna-steppe to woodland formations) in Brazil. This species begins to flower and produce fruit at an early age, usually at around two years. The seeds are dispersed by gravity (barochory) (Carvalho 2006). The wood of *M. caesalpiniifolia* can be used commercially as saw timber or round wood, as firewood, in landscaping, live fences, and beekeeping, as a medicinal plant, and for environmental recovery and restoration plantations (Carvalho 2006). The adaptability and utility of *M. caesalpiniifolia* for rehabilitating degraded soils has been demonstrated in other studies (Costa et al. 2004) and is considered to be a useful candidate species for facilitating early successional processes (Santana et al. 2009). Although the litter produced by this species has a high nutrient content and high decomposition rate (Campoe and Engel 2004; Fernandes et al. 2006), the potential allelopathic effect of its litter on native seed germination should be considered (Piña-Rodrigues and Lopes 2001).

We tested the following two hypotheses: (1) Removal of *M. caesalpiniifolia* will result in greater native woody species regeneration as measured by species richness and diversity, individuals density, stand basal area, and less infestation from weedy species. (2) The structure and composition of the regenerating community will change: light-demanding species will be the most benefited by the management intervention, and the management effects will be more evident in denser plantation systems and on sites with less fertile soils.

Materials and Methods

Study Area

The study area is located in Botucatu, south-central region of São Paulo State, Brazil (22°52′32″S and 48°26′46″W). According to Köppen, the climate is classified as Cfa

(Alvares et al. 2013), subtropical, wet, and hot. Annual rainfall averages 1494 mm with the rainy season lasting from October to March. Annual mean temperature is 20.5 °C, with the minimum average occurring in July and maximum in February. The natural vegetation is semideciduous tropical forest within the Atlantic Forest biome range.

Experimental Design

We conducted our research at two sites with contrasting soil fertility and landscape context. In Site 1, the soil is a fertile loamy Ultisol, with an elevation of 700 m. It is surrounded by farmlands and was about 50 m away from a disturbed riparian forest. Site 2 is characterized by relatively low fertility and sandy Alfisol soil, with an elevation of 547 m. It is located 100 m away from a late-successional remnant forest.

At each site, a randomized block design with three blocks and two restoration systems (mixed plantation using commercial timber and firewood tree species (MIX) and a modified "taungya" agroforestry system (AF)) was set up in January/February 1998. The two plantation systems were established in three replicate 50 × 50 m plots at each site. The AF system included mixed species tree plantings established in triple rows (spacing 1.5 m × 2 m) interspaced by 5 m width crop alleys where annual crop species were cultivated until canopy closure. Within the triple rows, the two outer rows consisted of 10 leguminous multipurpose fast-growing tree species (including *M. caesalpiniifolia*) that were selected for their utility as firewood (Online Appendix 1). In the inner rows, 10 slow-growing commercially valuable timber species were planted. After canopy closure (in 2005–2006), native fruit trees and medicinal plants were introduced, respectively, at site 1 and 2 within the former crop alleys. The other plantation system (MIX) consisted of a mixture of commercial timber and firewood tree species alternating rows with 10 fast-growing firewood species and 15 slow-growing timber species (Online Appendix 1). This system was similar to the AF system, but without the 5 m crop alleys.

The *M. caesalpiniifolia* management experiment was established between August 2011 and January 2012 (14 years after the plantation system establishment), using two levels (managed or control). The management levels were nested within the plantation system plots, resulting in four treatment combinations (plantation system x management level) and six replicates per experimental site (Fig. 1).

Description of the Experimental Management Techniques Adopted

The removal of *M. caesalpiniifolia* was initially planned to be carried out around the sixth year after the establishment of the restoration experiment. However, at that time, we found through studies of growth and biomass production (see Campoe and Engel 2004) that this species still had the potential for further biomass and firewood production. Furthermore, mainly due to the deposition of a thick litter layer and shading, this species was still playing an important weed suppression role in those systems. Therefore, it was decided to keep this species in the system for a longer time. However, 8 years later, with canopy closure and decreased weed infestation inside the experimental plots, their ecological role seemed to have decreased. Thus, we decided to experimentally remove this species. It is important to note that no other management practices were applied at that time.

Management treatments were applied in a split-plot design within the main plots. The latter were subdivided into four units of 25 × 25 m where the two management levels (managed: *M. caesalpiniifolia* trees eliminated, and control: *M. caesalpiniifolia* retained) were randomly assigned (Fig. 1).

As the wood harvested was supposed to be used as firewood, we standardized the harvest to include all *M. caesalpiniifolia* individuals higher than 1.30 m, in the managed subplots (25 × 25 m) (Fig. 1). The proportion of felled trees represented on average 20 % of the remaining planted trees at the time of management application. The canopy cover reduction soon after management ranged from 10 to 51 % (measurements made using an AccuPAR LP-80). The felled trees DBH ranged from 1 to 72 cm with an average of 8 cm.

To reduce the impacts of harvesting on the natural regeneration community, tree crowns were first pruned and then directionally felled to avoid damage to the residual trees and regeneration within the plots. Immediately after cutting, each stump was brushed with pure glyphosate. Six months later, resprouting trees were sprayed with glyphosate (Roundup 6 %). The wood of the trees harvested from the treatment subplots was removed to outside the plots and its volume measured in stacked meters.

Sampling and Data Collection

All data described below were collected within the study plots at three points in time: (a) immediately prior to (t_0), (b) two months after (t_2), and (c) one year after (t_{12}) the felling and removal of *M. caesalpiniifolia*. In each of the main plots (50 × 50 m), t_0 data collection was confined to the central 30 × 30 m area, which was divided into four subplots of 15 × 15 m. For all other measurements (t_2 and t_{12}), subplots measuring 10 × 10 m were sampled, leaving a 10-m buffer strip between them in order to avoid the influence of one treatment over the other (Fig. 1).

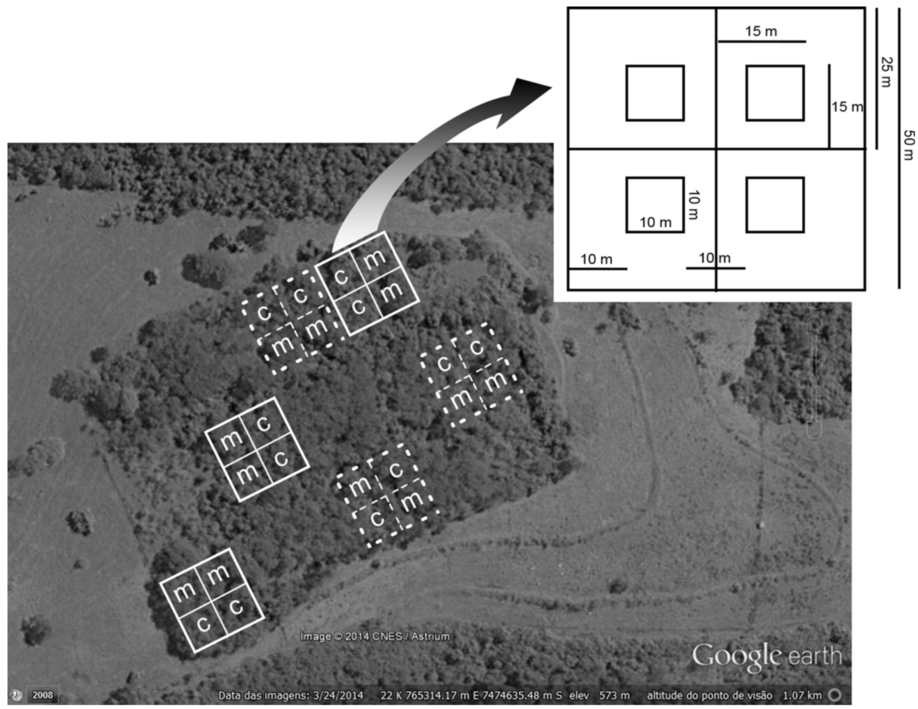

Fig. 1 Main plots and management units at site 2. *Dashed line*: main plots of MIX; *solid line*: main plots of AF; *c*: control; *m*: managed. Boundaries of: main plot (50 × 50 m); management unit (25 × 25 m); subplots sampled at t_0 (15 m × 15 m); subplots sampled at t_2 and t_{12} (10 × 10 m)

Natural Regeneration Structure and Composition

All trees and large shrubs were censused, identified to species, and categorized into two height classes, class 1: 0.20– <1.30 m; class 2: ≥1.30 m. Individuals that were identified as having been planted (when the plantation systems were established in 1997–1998) were not included in this analysis. Species were identified in the field when possible and vouchers specimens were collected for later identification by comparing them with specimens or through consultation with experts. The botanical classification system used was APG III and nomenclature followed the List of Species of the Brazilian Flora (2014). Class 1 individuals were only counted and identified. Class 2 individuals were counted, identified, and their diameters at breast heights (DBH) and height were registered.

Weed (Exotic Invasive Grasses) Cover

The 10 × 10 m subplots were divided into an 8 × 8 grid of 1.25 × 1.25 m quadrats. Each quadrat was visually inspected and classified into two classes of grass cover: <50; ≥50 %. The total grass cover was determined by the proportion of quadrats with more than 50 % of grass coverage in each subplot.

Data Analysis

Four variables commonly used to assess plant communities were calculated (individual's density, plot basal area, plot species richness, and species diversity) plus weed infestation (invasive exotic grass cover, % of area). Community metrics (density, species richness, and Fisher's alpha) were calculated following standard procedures (Magurran 2004; Newton 2007). The basal area calculation included only class 2 individuals from the natural regeneration community, as this metric is based on diameter at breast height (DBH, 1.30 m above ground). For all other community metrics, both natural regeneration classes (1 and 2) were pooled together.

To assess the effect of management on the regenerating community, a four-factor split-plot nested analysis of variance (ANOVA) was carried out for each of the five response variables. The two whole-plot factors were site (site 1 and site 2) and plantation system (MIX and AF). The split-plot factors were management (managed and control) and measurement time (t_2 and t_{12}). The subplot basal area of planted trees before management (measured in t_0) was added in the model as covariate in order to improve the control of pre-existing differences among plots, which are expected to exist after 14 years of establishment of the restoration systems. The analyses were carried out using the PROC MIXED procedure of SAS 9.0 (SAS Institute INC. 2002). When necessary, data were transformed to help meet assumption of normality and the factor site was added as a GROUP effect to increase the flexibility to the model heterogeneity in the covariance parameters. The most parsimonious models were determined by sequential backward removal of non-significant terms based on conditional F tests of ANOVA terms. Due to significant interactions terms involving management and measurement time, it was necessary to analyze management effect for each measurement time separately. To compare each management level into each measurement time, Bonferroni-corrected tests were performed ($P \leq 0.05/6$ tests $= 0.0083$) (Sokal and Rohlf 1995).

To evaluate changes in species composition affected by management, we performed sequentially non-metric multidimensional scaling analysis (NMDS) using Bray–Curtis index and cluster analysis with the unweighted pair group method with arithmetic mean (Kent and Coker 1992), using a matrix of abundance of individuals for each species and a composition of factors (2 levels of measurement time + 2 levels of site + 2 levels of plantation system + 2 levels of management). Both analyses were performed in R version 3.1.2 (R Core Team 2014) with package vegan 2.2-1 (Oksanen et al. 2015).

To compare the density of each species among measurement times (t_0, t_2, and t_{12}) within control plots and within managed plots, we performed the t test and then the Bonferroni-corrected test ($P \leq 0.05/3$ tests $= 0.0167$).

Results

Composition and Structure of Natural Regeneration Community

Considering all size classes of the natural regeneration pooled together, we found 10,856 individuals per hectare at site 1 before management (t_0) (8693 ind. ha^{-1} in the AF system and 13,019 ind. ha^{-1} in the MIX system). At site 2, there were 8841 individuals per hectare (5474 ind. ha^{-1} in the AF system and 12,207 ind. ha^{-1} in the MIX system). Ninety-seven species were identified at site 1, belonging to 36 botanical families, and 67 species were identified at site 2, belonging to 27 botanical families (Online Appendix 2). Soon after (t_2) the management, there were 16,133 individuals per hectare at site 1 (14,233 ind. ha^{-1} in the AF system and 18,033 ind. ha^{-1} in the MIX system), and 7604 individuals per hectare at site 2 (6242 ind. ha^{-1} in the AF system and 8967 ind. ha^{-1} in the MIX system). Seventy-seven species were identified at site 1, belonging to 33 botanical families and 61 species were identified at site 2, belonging to 29 botanical families (Online Appendix 2). One year after the management treatment was applied (t_{12}), there were 16,013 individuals per hectare at site 1 (12,042 ind. ha^{-1} in the AF system and 19,983 ind. ha^{-1} in the MIX system). At site 2 there were 10,717 per hectare (7817 ind. ha^{-1} in the AF system and 13,617 ind. ha^{-1} in the MIX system). Eighty-eight species were identified at site 1, belonging to 34 botanical families, and 70 species were identified at site 2, belonging to 32 botanical families (Online Appendix 2).

The NMDS (stress = 0.091; non-metric fit, $R^2 = 0.992$; linear fit, $R^2 = 0.968$) and cluster analysis of species data (Fig. 2), indicated two distinct groups of plots. The first group, more cohesive, comprised all plots at site 1 and the second group, less cohesive, comprised all plots at site 2.

Species Richness and Diversity of Natural Regeneration Community

Both observed species richness and Fisher's alpha species diversity were affected significantly by site and time after management. The Fisher's alpha diversity was significantly affected by the interaction effect of management x time after management (Tables 1, 2). Otherwise, neither species richness nor Fisher's alpha diversity were significantly affected by plantation system. Mean species richness and Fisher's alpha were greater at site 1 ($S = 26$, $\alpha = 9.2$) than at site 2 ($S = 13$, $\alpha = 5$). Furthermore, they increased from $S = 18$ and $\alpha = 6.4$ in t_2 to $S = 21$ and $\alpha = 7.8$ in t_{12}. The Fisher's alpha of managed plots increased from t_2 to t_{12} (Fig. 3). Nevertheless, there were no statistical differences between managed and control plots in t_{12} measurements.

Density of the Natural Regeneration Community

The density of natural regeneration was significantly affected by time of measurement, by plantation system, and

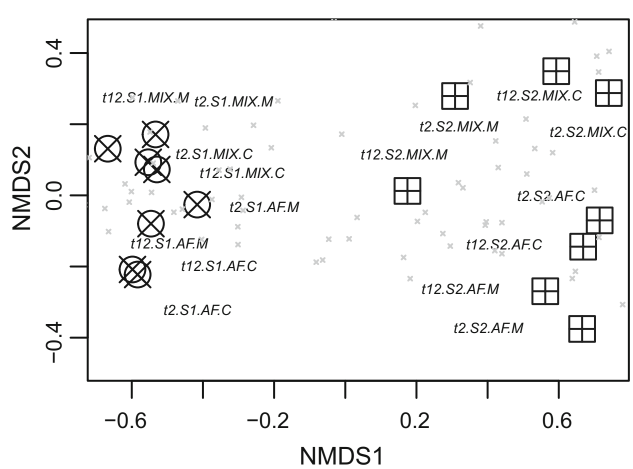

Fig. 2 Non-metric multidimensional scaling (NMDS) using Bray–Curtis index and a matrix of abundance of individuals for each species and a composition of factors (measurement time + site + plantation system + management level). Stress = 0.091; non-metric fit, $R^2 = 0.992$; linear fit, $R^2 = 0.968$. t_2: soon after management; t_{12}: one year after management; *S1*: site 1, *S2*: site 2, *MIX*: mixed plantation using commercial timber and firewood tree species; *AF*: modified "taungya" agroforestry system; *M*: managed, *C*: control, × Species, ⊗ Group 1, ⊞ Group 2. Groups achieved from cluster analysis with the unweighted pair group method with arithmetic mean (UPGMA)

Table 1 Results of analysis of variance for treatment effects on density, basal area, number of species, Fisher's alpha diversity index, and weed infestation

Source of variation	Density (ind. ha^{-1})		Basal area (m^2 ha^{-1})		Number of species (S)		Fisher's alpha (α)		Weed infestation (%)	
	F	*P* value	*F*	*P* value	*F*	*P* value	*F*	*P* value	*F*	*P* value
Management			4.17	0.0488					14.38	0.0006
Plantation system	5.75	0.0250	12.11	0.0059						
Site					43.14	0.0028	52.12	0.0020		
Time after management	10.57	0.0021	37.84	<0.0001	32.20	<0.0001	33.18	<0.0001	37.00	<0.0001
Management × time after management			12.73	0.0009			4.24	0.0195		
BA_t_0	5.57	0.0230								

F values: Blank = $P > 0.05$ (not considered in the model). BA_t_0: Basal area of planted trees before management, used as covariate

Table 2 Mean values for density, basal area, number of species, Fisher's alpha diversity index, and weed infestation in each level of the source of variation

Source of variation	Density (ind. ha^{-1}) Mean	Basal area (m^2 ha^{-1}) Mean	Number of species (S) Mean	Fisher's Alpha (α) Mean	Weed infestation (%) Mean
Management					
Managed	11,915 a	1.18 a	20 a	7.6 a	30 a
Control	13,319 a	1.92 b	18 a	6.6 a	17 b
Plantation system					
AF	10,083 a	0.85 a	18 a	7.2 a	27 a
MIX	15,150 b	2.25 b	21 a	7.0 a	20 a
Site					
1	16,073 a	1.51 a	26 a	9.2 a	19 a
2	9160 a	1.59 a	13 b	5.0 b	28 a
Time after management					
t_2	11,869 a	1.26 a	18 a	6.4 a	17 a
t_{12}	13,365 b	1.84 b	21 b	7.8 b	30 b

Lower case letters indicate differences between means (α = 0.05)

by the covariate BA_t_0 (Tables 1, 2). Otherwise, it was not affected by the management, by sites, and by any interaction. Mean density increased significantly from 11,869 ind. ha^{-1} in the t_2 measurement to 13,365 ind. ha^{-1} in the t_{12} measurement. Mean density was greater in the MIX (15,150 ind. ha^{-1}) than in the AF (10,083 ind. ha^{-1}).

The mean plant density of each species between measurement times (t_0 and t_{12}) showed a different pattern of response to the management: neutral, negative, and positive (Fig. 4). A neutral response was observed for the large majority (90 %) of species as the same pattern of variation in density was observed in both levels of management (managed and control). In other words, an increase, decrease, or no change in density occurred similarly in both management levels from one year to the other. A negative effect of the management (stability of plant density in the control plots and decrease in the managed plots) was observed for 7 % of the species (*Psidium guajava* L., *Zanthoxylum caribaeum* Lam., *Schinus terebinthifolius* Raddi, *Allophylus edulis* (A.St.-Hil. et al.) Hieron. ex Niederl., and *Celtis fluminensis* Carauta). A positive effect of the management (increased density in managed plots and no change in control plots) was observed for 3 % of the species (*Citharexylum myrianthum* Cham. and *Pterogyne*

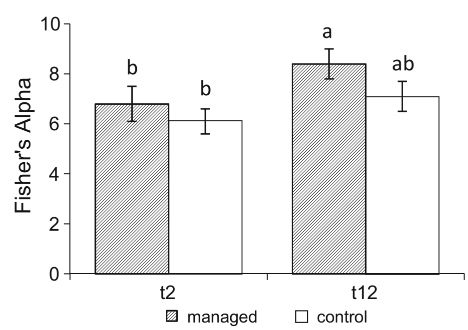

Fig. 3 Effect of management on the mean (±1 SE) Fisher's alpha diversity soon after (t_2) and 1 year after (t_{12}) the management of plots (elimination of *M. caesalpiniifolia*)

nitens Tul.). There was no significant difference for any species between t_2 and t_{12}.

The percentage of *M. caesalpiniifolia* individuals in natural regeneration in t_{12} was equal to or less than their percentage in t_0, for all samples (Online Appendix 2).

Basal Area of Natural Regeneration Community

Plot basal area was significantly affected by management, by plantation system, by time of measurement, and by the interaction effect of management × time of measurement (Tables 1, 2). Basal area was not affected by site. The interaction resulted from the significant management effect being detected in the t_2 but not in the t_{12} measurements (Fig. 5), as seen the basal area was significantly lower in the managed plots t_2 but there was no difference between managed and control in the t_{12}. Mean basal area was greater in the MIX (2.25 m^2 ha^{-1}) than in the AF (0.85 m^2 ha^{-1}) (Table 2).

Weed Infestation

Weed infestation was significantly affected by management and by time of measurement, but was not affected by plantation system, site, nor by any interaction (Tables 1 and 2). Mean grass cover was higher in the managed plots (30 %) compared to the control plots (17 %) and increased from 17 % in the t_2 to 30 % in the t_{12} measurements. The predominant invasive grass was *Megathyrsus maximus* (Jacq.) B.K. Simon & S.W.L. Jacobs at site 1, and *Urochloa decumbens* (Stapf) R.D. Webster at site 2.

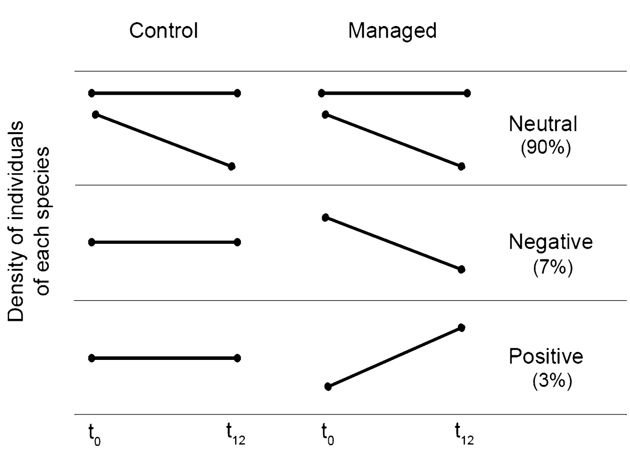

Fig. 4 Effect of management on plant density of each species between measurement times, before (t_0) and one year after (t_{12}) management, within control plots and within managed plots (elimination of *M. caesalpiniifolia*). Comparisons were made performing *t* test and then Bonferroni-corrected test ($P \leq 0.05/3$ tests $= 0.0167$) for each species and then the species were grouped by their response, neutral, negative, or positive

Discussion

Effect of Management

Despite the use of low impact tree harvesting techniques, the damage to the regenerating community resulting from tree felling and subsequent wood removal was very high, as shown by the basal area decrease observed soon after management. However, this loss was quickly recovered and within one year the basal areas in the treatment plots equaled those in the control plots (Fig. 5). Apparently, the intensity of competition decreased with thinning and the effect of the established community on the natural regeneration growth reduced. This was also noted by Wilson and Tilman (1993) studying grasslands in Minnesota, USA. The increase in growth in response to disturbances of different scales is a common pattern in woody plants (Canham and Marks 1985). According to ecological filters theory, this may be explained both by the effect of the disturbance caused by the management, that can act as an ecological filter itself (Hobbs and Norton 2004), as well as by the canopy trees acting as an ecological filter for those plants that attempt to regenerate under them (Gandolfi et al. 2007). Therefore, the *M. caesalpiniifolia* removal itself may have affected the community assembly and/or the new pool of canopy tree species after management may determine, in some extent, the community structure and composition.

As a result of the openings created by tree felling, environmental filters might have changed, and as an expected consequence the light availability increased (Belote et al. 2012), resulting in higher weed infestation. Weeds are a significant management issue that often hinders restoration projects (D'Antonio and Meyerson 2002). However, in this study, we observed that the increased grass cover did not cause serious damage in the regeneration process, since both species diversity and basal area increased, and species richness and plant density of the

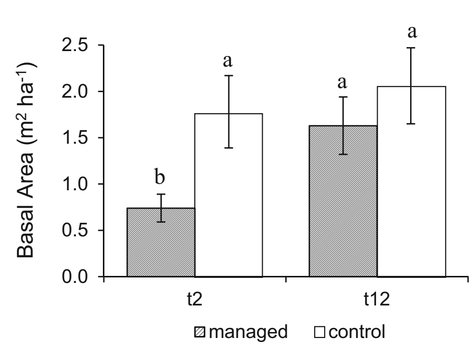

Fig. 5 Effect of management on the mean (±1 SE) basal area soon after (t_2) and one year after (t_{12}) the management of plots (elimination of *M. caesalpiniifolia*)

managed community regeneration did not change. It is expected that with further canopy development, the grass cover will decrease progressively over time, and thereby reduce the risk of its inhibiting development of the regenerating community.

Our results indicate that management can affect individual species in different ways, as shown by the variation in the density of some species and by the slight changes in floristic composition observed soon after the management at site 2. As *M. caesalpiniifolia* is a light-demanding species, we expect as soon as the canopy closes, after the management, this species will be suppressed. However, during the study period, there was no predictable pattern in relation to a specific successional group of species, therefore suggesting a broad-spectrum and non-specific effect. This has been previously reported by Hubbell et al. (1999), studying canopy gaps in the tropical moist forest in Barro Colorado Island, Panama. Considering that disturbances are dynamic mechanisms that modify the process of community assembly through time (Jentsch 2007), monitoring this effect is imperative, since both positive effects-like the convergence of secondary forests to old-growth forests composition (Dupuy and Chazdon 2008) and negative effects, as the inertia or even successional retreat related to the species composition (Clark and Covey 2012) are expected to occur.

Although *M. caesalpiniifolia* may act as a filter, influencing community assembly rules, it is more likely that changes in community structure recorded in this study are due the management itself and associated changes in resource availability, promoting a continuous dynamics that can control the establishment and turnover of individuals and species, as postulated by White and Jentsch (2004). This was noticeable by the similar effects on regenerating community by management registered in different species (e.g. Ashton et al. 1998; Kueffer et al. 2010; Belote et al. 2012).

Effects of Site and Plantation System

In spite of the evidence of floristic dissimilarity between managed and control plots, it was found that the determinant factor driving species composition was the physical environment, represented in this study by the sites (maximum similarity of 0.12). Neither the plantation systems nor the management applied had significant effects. Supporting these findings, we can cite other studies that have shown variations in one or more environmental characteristics, such as slope steepness and position, annual rainfall, rainfall seasonality, soil fertility, acidity, and texture, among others which may have strong influence on growth (Baker et al. 2003; Henry et al. 2004; Park et al. 2010; Breugel et al. 2011), survival (Henry et al. 2004; Park et al. 2010), structure (Ashton 1988; Baker et al. 2003; Levula et al. 2003; Park et al. 2010), and species richness (Wright 1992; Clinebell II et al. 1995; Stevens and Carson 1999) of plant communities.

Regarding the effect of the physical environment over the regenerating communities, it was observed that species richness and diversity were higher at site 1 (the most fertile soil) compared to the site 2 (less fertile). It is interesting to note that this result contrasts with other studies such as Stevens and Carson (1999) and Henry et al. (2004) who found that species richness decreased according to the increase in soil fertility. Stevens and Carson (1999) concluded in their study that the lower density of individuals in more fertile soils can result in a smaller number of individuals sampled on the same size area, and consequently the lowest richness was only a sampling effect. Nevertheless, Henry et al. (2004) suggested that establishment limitation, such as the decrease in seedlings growth and survival, imposed by the established individuals can control the recruitment of species in high fertility soils. In our study, we found that there was neither a significant variation in density nor in plant growth between sites. Thus, the results suggest that the influence of the soil properties (fertility and water retention) on the succession processes/ecosystem restoration was stronger than that of the species pool from neighboring fragments.

Firewood Yield and Revenue from the Managed Plots

The mean amount of firewood obtained from the management either by plantation system ($F = 1.69$, $P = 0.2348$), or by site ($F = 1.21$, $P = 0.3084$) was 84 stacked meters per hectare.

The average trading price for fire wood from *M. caesalpiniifolia* in the local market was US$14.00 per stacked meter. With the average yield from management activity of 84 stacked meters per hectare, the revenue generated could reach US$ 1150.00 ha^{-1} as economic benefit.

Most land owners in Brazil are required by law to restore a portion of their land which is considered relevant for conservation purposes (see Aronson 2010). The majority of them, however, have no money to carry out the necessary restoration activities, which may include soil tillage, buying seedlings, and maintenance in the first years. Thus, the potential revenue generated by selling fire wood from the restored ecosystems under management may represent an extra revenue source to small farmers, helping them to offset restoration costs and facilitating the achievement of the restoration goals.

Strategies that incorporate knowledge about ecological processes and their application into the design of more efficient agricultural systems (productive and sustainable) represent a more suitable way to exploit natural resources in the meaning of ecological intensification (Malézieux 2012; Bommarco et al. 2013). The use of nurse plants (like *M. caesalpiniifolia*) in restoration projects is one of these strategies that can be used to favor the restoration of degraded ecosystems that have severe environmental filters preventing initial natural regeneration processes (see Badano et al. 2009; Ren et al. 2008; Padilla and Pugnaire 2006; Elliott et al. 2003; Ashton et al. 1997). However, the positive effect on restoration by a species included in the system (aiming to enhance their natural ecological processes) may not be permanent. The relationship between species and their role in community dynamics change with time. In this way, hypothetically, a species that poses positive effects on community development in the early successional process can cause negative effects later on. In case if this hypothesis be true, the species elimination is required within an adaptive management approach (sensu Borman et al. 1999). In this study, the removal of *M. caesalpiniifolia* appeared to have positive effects on the regeneration community in the short term (one year), but it should be noted that monitoring studies are still required to evaluate the longer term effects both of removal of this species and its retention in these restored forest systems. The assessment of tradeoffs involved in this strategy requires a more complete understanding of the effects of inclusion, permanence, and removal of the facilitative species in the system.

The effects of tree removal over regenerating communities are complex, as there is considerable unpredictability on ecosystem responses to management actions. It should be recognized that changes in resource availability caused by management will not always favor the target community, and in some cases may favor non-native species instead. Therefore, early identification of possible undesirable effects may help to define the best strategies to minimize the negative and maximize the positive effects of management.

Conclusion

Our study suggests that the management intervention (removal of *M. caesalpiniifolia*) facilitated the regenerating community and had no negative effect on it. The removal of the exotic species can be recommended, especially taking into account the possibility of economic revenue of forest ecosystems undergoing restoration. Although the removal of *M. caesalpiniifolia* did not bring great ecological benefits during the short period studied, such as the increase in species diversity, it certainly did not provoke damages. Thus the non-native species removal can be recommended as a management practice when seeking some revenue from firewood exploitation without compromising ecological succession in ecosystem restoration. Even if economic revenues from the plantation systems are not expected, the elimination of this exotic species is recommended to accomplish for ecological restoration goals and to prevent ecosystem invasion.

In this study, the basal area, species diversity, and grass infestation were more sensitive to the effects of management and appear to be the most useful response variables for monitoring management effects. These variables should be used as part of monitoring program of managed ecosystems undergoing restoration.

Though the results of this study suggest removal of *M. caesalpiniifolia* may benefit regenerating woody species and possibly accelerate succession, conclusions about the effect of tree removal as a management strategy will depend on further monitoring studies to confirm the beneficial long-term effects.

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