

THE ROLE OF *BACCHARIS* (ASTERACEAE) SHRUBS IN THE SHORT-TERM RESTORATION OF ATLANTIC RAINFOREST

Walisson K. Siqueira¹ , Arthur L. Moura² , Ramón Perea³ ,
Michel J. P. Alves⁴ , Geraldo W. Fernandes¹ 

¹Universidade Federal de Minas Gerais, Brazil
e-mail: kenedy.siqueira@gmail.com, gw.fernandes@gmail.com

²Auburn University, USA
e-mail: arthurlamouniermoura@gmail.com

³Universidad Politécnica de Madrid, Spain
e-mail: ramon.perea@upm.es

⁴Universidade Federal Do Pará, Brazil
e-mail: micheljacobypereiraalves@gmail.com

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The introduction of nurse species on degraded sites of Permanently Protected Areas represents a useful strategy for vegetation recovery in riparian forest. Species of the genus *Baccharis* (Asteraceae) have been documented as potential nurse plants being able of restructuring the native plant community. This study was aimed to evaluate the potential nursing role of *Baccharis dracunculifolia* in the recovery of a native plant community of a degraded Atlantic rainforest in a short-time period. The study was developed in two abandoned pasture areas in a riparian forest in the River Piranga basin, one in which *B. dracunculifolia* was planted (Restored treatment) to promote restoration, and a second area with no intervention (Degraded = Control treatment). Sampling took place 18 months after the planting of *B. dracunculifolia*. We set up 20 plots of 2 × 2 m in each treatment type (n = 40 plots), where all plant species were recorded (with the exception of the introduced *Baccharis* individuals and grasses), classifying them as native, ruderal, or alien. Plant richness was twice higher in the Restored treatment than the Degraded treatment. Furthermore, the observed values of alpha, gamma and beta diversity were also higher in Restored treatment. Restored treatment had 17 exclusive native species, while the Degraded treatment had only three non-exclusive native species. In addition, fewer ruderal and alien species were recorded in the restored plots with *B. dracunculifolia* compared to degraded plots. We concluded that, even in a short time period, planting *B. dracunculifolia* had a positive effect on promoting the assembly of the native plant community and possibly decreasing the chances of invasion by alien species.

Key words: community assembly, ecological restoration, null model, plant facilitation, plant invasion, River Piranga basin, species colonisation

Introduction

The structure and composition of plant communities is strongly influenced by the physical, chemical and biological conditions of the environment. Degraded environments, such as over-exploited pastures, abandoned mining areas, and deforested riparian forest usually lead to a poor flora, often dominated by ruderal or even alien species (Chaneton & Facelli, 1991; Carvalho et al., 2014). In these degraded environments, stressful physical conditions and a lack of nutrients limit the establishment of native species adapted to the natural habitat (Bradshaw & Chadwick, 1980; Perrow & Davy, 2002). Generally, for the restructuring of plant communities in degraded areas spontaneous succession or some restoration measures can be used (Holl & Aide, 2011; Chazdon & Uriarte, 2016; Le Stradic et al., 2018).

The main interventions for environmental recovery in degraded areas aim to restructure the abiotic conditions that enhance local plant colonisation (Parrotta & Knowles, 1999; Corrêa, 2009; Holl & Aide, 2011). The spontaneous succession by plant colonisation occurs through the propagules that come from nearby patches of native vegetation or directly from the soil seed bank (Bakker et al., 1996; Crouzeilles et al., 2017). Planting or direct seeding of native species favouring the colonisation of other plant species have been considered one effective measure to accelerate the colonisation and recovery of degraded areas (Cicczazzo et al., 2014; Spadeto et al., 2017). These plants having a positive effect on the establishment, survival and growth of other plants are known as nurse species (Franco & Nobel, 1989; Cavieres & Badano, 2010;

Holmgren & Scheffer, 2010). Thus, the «nurse» or «benefactor» plants facilitate the development of other plant species («beneficiary») by improving the physical conditions of the habitat (e.g. temperature, humidity, and availability of organic matter in the soil; see Ren et al., 2008; Li et al., 2017; Lu et al., 2018). This facilitative interaction has a practical side when applied to ecological restoration as the nurse species may provide shelter for the native target species (Pardilla & Pugnaire, 2006). The use of effective nurse plants is particularly relevant in highly compacted soils and in mined landscapes where restoration is imperative (e.g. Fernandes, 2016).

The use of nurse species in ecological restoration projects has become of great relevance as they aid in community assembly in early stages of natural succession and result in increased richness and abundance of other species (e.g. Kikvidze et al., 2005; Spadeto et al., 2017; Fagundes et al., 2018; Shaw, 2018). Additionally and importantly, some nurse plants may create microsites capable of restoring the native plant community in areas highly invaded by alien plant species (Brennan et al., 2018; Perea et al., 2019). This is of particular relevance in ecosystems where invasive species are widespread and where restoration projects have failed due to biological invasion, such as in the Cerrado and Atlantic rain forest of Brazil (e.g. Fernandes et al., 2015).

Many pioneer species play a very important role in the process of natural succession, and are used in the early colonisation of recovering environments (Cao et al., 2011). Herbaceous plants with rapid growth, high reproductive capacity and dispersion, and adapted to disturbed areas are generally called ruderal species (Grime, 1977). Often, ruderal species may persist in degraded areas when the disturbance is recurrent or occurred in great magnitude (Grime 1977; Řehouňková & Prach, 2008). Thus, the use of nurse species in degraded areas could also alleviate the effects of persistent ruderal species in the plant community (Parrotta et al., 1997).

The genus *Baccharis* L. (Asteraceae) has been appointed as nurse of other species (Callaway & D'Antonio, 1991; Gómez-Aparicio et al., 2004; Sánchez-Velásquez et al., 2011; Peláez et al., 2019). For instance, field experimental tests indicate that *Baccharis dracunculifolia* DC. has the potential to be a nurse species in degraded environments of montane Neotropics

(Perea et al., 2019). *Baccharis dracunculifolia* is a dioecious shrub with characteristics of a pioneer plant, such as fast growth and development (Negreiros et al., 2014). Rapid growth rates enable soil protection against direct radiation exposure, either through the shading provided by its branches, or by the covering of organic matter (Tiedemann & Klemmedson, 1977). In addition, *B. dracunculifolia* provides a large amount of ecosystem services, including a rich fauna of insects and pollinator species, which are important to ecosystem functioning and sustainability (Fernandes et al., 2014, 2018).

The goal of this study was to experimentally evaluate the effects of planting *B. dracunculifolia* in a degraded pasture in a soap stone mining company in the River Doce basin as a possible nurse species. Therefore, by planting *B. dracunculifolia* in a degraded area we expected that the nurse species would facilitate habitat recovery both by aiding in the establishment of native species and by decreasing the colonisation by ruderal and alien species. The following hypotheses were tested: i) the plant diversity is higher in the area planted with *B. dracunculifolia* compared to a degraded area; ii) the colonisation by native plant species (richness and abundance) is higher in the area planted with *B. dracunculifolia* compared to a degraded area; iii) less ruderal native and alien plants are found in the area where *B. dracunculifolia* was planted compared to a degraded area.

Material and Methods

Study area

The study was carried out in the River Piranga basin, a tributary of the River Doce, in Minas Gerais, Brazil. The entire region is known for its high land use change from rain forest to cattle ranching, eucalypt plantation, and some mining (Gomes & Maciel, 2018). Similarly, the study area (20.67108° S, 43.3733° W) was part of a continuous riparian forest in the River Piranga basin, transformed into a pasture and abandoned ten years, dominated by the African grass *Uruchoa* spp. before this study took place. In addition to be protected by the Brazilian Atlantic Forest Law (Law 11.428/2006), riparian ecosystems are considered Permanently Protected Areas (PPA) according to the Brazilian forest code (Law 12651/2012) (Metzger et al., 2019).

The first site is an area of abandoned pastures (ten years of successional age), adjacent

to an area of deposition of sterile soapstone material of proximity about 250 m² (11 × 23 m). Subsequently (January 2017), the area was restored using solely 200 saplings of *B. dracunculifolia* of 20–25 cm height (hereinafter – Restored treatment) (Fig. 1A). Saplings were introduced in the area by direct planting at an average distance of 2.0–2.5 m from each other. The second site is also a pasture area that

did not receive a recovery intervention and it is, therefore, an abandoned area after ten years as a pasture (hereinafter – Degraded treatment) (Fig. 1B). Both sites were bordered by native Atlantic Forest remnant vegetation (ca. 10 m away). In both environments, species colonisation was let to occur naturally during the experiment. No other intervention was occurred during the experiment.

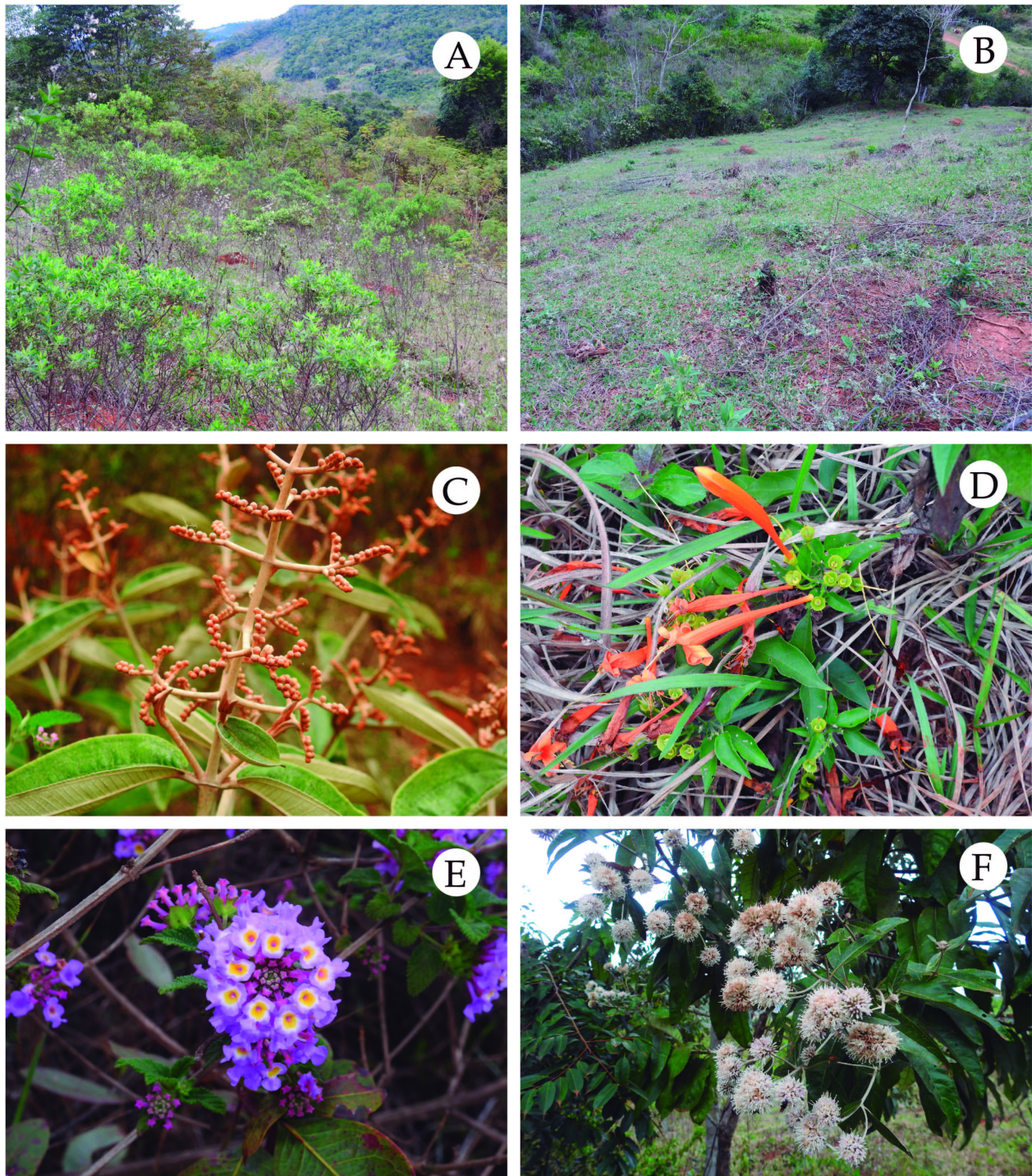


Fig. 1. Restored treatment (A) and Degraded treatment (B) of riparian forest in the River Piranga basin, Brazil. Some sampled species: *Miconia albicans* (C), *Pyrostegia venusta* (D), *Lantana camara* (E), *Eremanthus erythropappus* (F). (Photos: Geraldo W. Fernandes).

Field sampling

Sampling took place in September 2018, one and a half year after the planting of *B. dracunculifolia* in the Restored treatment. The species that colonised the Restored and Degraded treatments were counted and identified in 20 plots of 2.5×2.5 m (i.e. 6.25 m^2 each, 125 m^2 in total) randomly distributed in each treatment, except for grasses, because the matrix of grass was already present before the beginning of the experiment. In the Restored treatment each vertex of every plot had a *B. dracunculifolia* individual (not computed in the counting). All the plants sampled were pressed in the field and later identified to the lowest possible taxonomical level by botanical specialists. In addition, the species nomenclature was corrected and updated using the function Plantminer of the Taxize package (Chamberlain & Szöcs, 2013) on R software (R Core Team, 2020).

Classification of the species sampled

In this study, the sampled species were classified according to the following status: native, ruderal or alien. Native species are species that naturally occur in Brazilian biomes (Zappi et al., 2015). Ruderal species are widely distributed native species, frequently documented in disturbed environments (Grime, 1977; Fernandes et al., 2015). Alien species are non-native species of Brazil, including naturalised exotic species (see Moro et al., 2012).

Statistical analyses

All analyses were performed in the R software (R Core Team, 2020). Initially, we evaluated separately the plant diversity on each site (Restored and Degraded) utilising metrics of diversity partition. Therefore, the diversity partition (alpha-, gamma-, and beta-diversity, respectively: (α) local, (γ) regional diversity and (β) species composition similarity) was calculated using the Entropart package (Marcon & Hérault, 2015) from the Simpson index obtained using the vegan package (Oksanen et al., 2013). The observed diversity values were compared with simulated communities created through null models that were pondered using the species abundance of the original composition matrix (e.g. Bogoni et al., 2017). Each null model was randomised 1000 times, and the observed results were compared with the expected results (p-value significance < 0.05), comparing the β -diversity in the null models. A multiplicative framework was used in all of the diversity analyses.

The hypotheses about plant diversity and colonisation by native plant species between treatments were evaluated through the construction of generalised linear mixed models (GLMM) using the lme4 package of R software (Bates et al., 2015), with an error distribution to each model (Crawley, 2013). In this case, the explanatory variable «site» was used as a random effect in all of the constructed models. Thus, only the effect of the nurse species could be observed and was tested through an analysis of variance (ANOVA) and, whenever necessary, a contrast analysis was done between levels of ANOVA (Crawley, 2013).

The first hypothesis predicts a greater diversity of species in the Restored than in the Degraded site. This was evaluated using the treatments («Restored» vs. «Degraded») as the explanatory variables. Therefore, the Simpson diversity index with Gaussian distribution of errors and the richness of plants with Poisson distribution of errors were used as response variables. In this case, the Simpson index was calculated for each plot. Additionally, the diversity partitioned results of the simulated communities were also tested: α -, γ -, and β -diversity with Gaussian distribution of errors were used as response variables.

The second hypothesis was evaluated through two predictions. To test the prediction that the richness of native plants is higher than ruderal and alien plants in the Restored treatment compared to the Degraded treatment, we used a GLMM. The treatments («Restored» vs. «Degraded») were used as explanatory variables, while plant species coverage (in percentage within each group, «native» and «alien») were used as response variables. In this case, the binomial error distribution was the most adequate to compose each model (Crawley, 2013). To test the prediction that the abundance of native plants is higher than alien plants in the Restored environment compared to the Degraded treatment, we used another GLMM. For this, the treatments («Restored» vs. «Degraded»), the status of species classification («native» vs. «alien»), and the interaction between these variables and treatments were used as explanatory variables. In this case, the abundance of plants, with Poisson distribution of errors (Crawley, 2013) was used as the response variable. The evaluation of the third hypothesis follows the same criteria and variables of test (explanatory and response) used in the second hypothesis, differing only in the use of the variable status of species classification. In

this case, the category «native» was divided in two, namely ruderal native plants (heretofore – «ruderal») and non-ruderal native plants (heretofore – «native»).

Results

A total of 1523 individuals, 1062 in the Restored site and 461 in the Degraded site, were

sampled. These individuals belong to 32 species of plants, being 29 species recorded in the Restored site and 11 species recorded in the Degraded site (Table 1). In the Restored site, approximately twice the number of species was sampled in relation to the Degraded site (Table 1), among which four were ruderal species, 17 were native and none were alien (Table 1).

Table 1. List of species sampled in Restored treatment (area with intervention for recovery) and Degraded treatment (area without intervention for recovery) of riparian forest in the River Piranga basin, Brazil, as to the classification (status) and the frequency of total occurrence of each species in the treatments

Species	Family	Status	Restored environment (%)	Degraded environment (%)
<i>Aspidosperma australe</i> Müll.Arg.	Apocynaceae	Native	0.37	0
<i>Achyrocline satureioides</i> (Lam.) DC.	Asteraceae	Ruderal	0.28	0
<i>Ageratum fastigiatum</i> (Gardner) R.M.King & H.Rob.		Ruderal	0	1.08
<i>Baccharis dracunculifolia</i> DC.		Native	0.84	0
<i>Baccharis serrulata</i> (Lam.) Pers.		Native	1.41	0
<i>Baccharis trimera</i> (Less.) DC.		Native	0.09	0
<i>Chromolaena odorata</i> (L.) R.M.King & H.Rob.		Ruderal	3.77	0
<i>Cyrtocymura scorpioides</i> (Lam.) H.Rob.		Native	3.58	0
<i>Eremanthus erythropappus</i> (DC.) MacLeish		Native	1.03	0
<i>Vernonanthura phosphorica</i> (Vell.) H.Rob.		Ruderal	2.82	28.41
<i>Vernonia</i> sp Schreb		Native	0.66	0
<i>Handroanthus ochraceus</i> (Cham.) Mattos	Bignoniaceae	Native	0.09	0
<i>Pyrostegia venusta</i> (Ker Gawl.) Miers		Native	1.13	0
<i>Stizophyllum perforatum</i> (Cham.) Miers		Native	0.18	0
<i>Croton glandulosus</i> L.		Ruderal	9.51	5.20
<i>Mabea fistulifera</i> Mart.		Native	13.47	0
<i>Chamaecrista cipoana</i> (H.S.Irwin & Barneby) H.S.Irwin	Fabaceae	Ruderal	1.03	4.77
<i>Desmodium barbatum</i> (L.) Benth.		Alien	1.60	3.25
<i>Piptadenia gonoacantha</i> (Mart.) J. F. Macbr.		Native	4.14	0
<i>Plathymenia foliolosa</i> Benth.		Native	1.88	0
<i>Stylosanthes guianensis</i> (Aubl.) Sw.	Ruderal	0	0.43	
<i>Sida cordifolia</i> L.	Malvaceae	Ruderal	25.63	48.15
<i>Waltheria indica</i> L.		Alien	5.37	0.21
<i>Miconia albicans</i> (Sw.) Steud.	Melastomataceae	Native	0.66	0
<i>Pleroma granulatum</i> D. Don		Native	0.37	0
<i>Psidium guajava</i> L.	Myrtaceae	Native	0.18	0
<i>Andropogon bicornis</i> L.	Poaceae	Ruderal	0.28	0
<i>Imperata brasiliensis</i> Trin.		Native	0.37	0
<i>Spermacoce verticillata</i> L.	Rubiaceae	Ruderal	4.52	0
<i>Serjania lethalis</i> A. St.-Hil.	Sapindaceae	Native	0	0.21
<i>Solanum lycocarpum</i> A. St.-Hil.	Solanaceae	Native	0.84	0.21
<i>Lantana camara</i> L.	Verbenaceae	Native	13.76	8.02

The observed results of the diversity partition in the Restored site were 3.82 for alpha-diversity and 8.15 for gamma-diversity (Fig. 2A). While in the Degraded site the observed results were 2.22 for alpha-diversity and 3.27 for gamma-diversity (Fig. 2B). The beta-diversity in both treatments was different to the expected by random ($p < 0.01$), with higher values for the expected (null) models for both the Restored treatment ($\text{Beta}_{\text{obs}} = 2.13$; $\text{Beta}_{\text{exp}} = 4.07$; Fig. 2C) and the Degraded treatment ($\text{Beta}_{\text{obs}} = 1.47$; $\text{Beta}_{\text{exp}} = 6.32$; Fig. 3D). In addition, in the simulated communities, the alpha-diversity was also larger in the Restored site ($\bar{x} = 5.65$, $\text{SE} = \pm 0.010$; Table 2) in relation to the Degraded site ($\bar{x} = 2.25$, $\text{SE} = \pm 0.004$; Table 2), as well as the gamma-diversity (Restored treatment: $\bar{x} = 22.95$, $\text{SE} = \pm 0.04$; Degraded treatment: $\bar{x} = 14.21$, $\text{SE} = \pm 0.08$; Table 2). On the other hand, the beta-diversity of the simulated communities was higher in the Degraded site ($\bar{x} = 6.32$,

$\text{SE} = \pm 0.03$; Table 2) in comparison with the Restored site ($\bar{x} = 4.07$, $\text{SE} = \pm 0.01$; Table 2).

The index of Simpson’s diversity was higher in the Restored site ($\bar{x} = 0.72$, $\text{SE} = \pm 0.005$) in comparison with the Degraded site ($\bar{x} = 0.69$, $\text{SE} = \pm 0.006$) ($p < 0.01$; Table 2, Fig. 3A). Furthermore, the plant richness of the Restored site was twice higher ($\bar{x} = 8.55$, $\text{SE} = \pm 0.49$) than in the Degraded site ($\bar{x} = 3.70$, $\text{SE} = \pm 0.27$) ($p < 0.01$; Table 2, Fig. 3B). Regarding the classification of species, the richness of native ($X_{(1, 240)} = 160.43$, $p = 0.5287$) and alien plants ($X_{(1, 240)} = 160.43$, $p = 0.5310$) were not different between treatments. On the other hand, when the ruderal natives are considered, the richness of native and ruderal plants differed according to the site, which did not occur with the richness of alien plants (Table 2). In the Restored site, the greater richness was mostly comprised of native species (about 50%, Fig. 4A), while in the Degraded environment, ruderal species were predominant (about 75%; Fig. 4A).

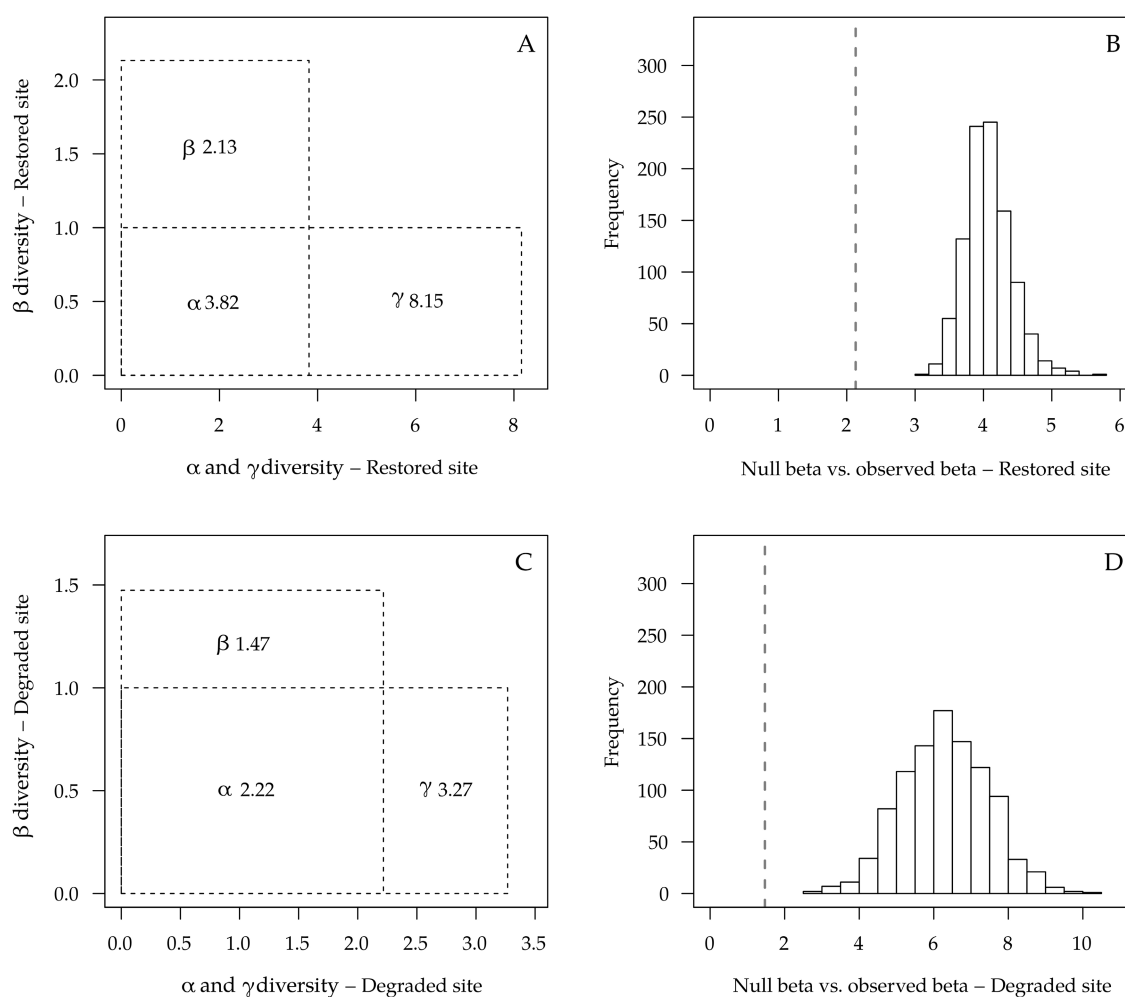


Fig. 2. Diversity partition (alpha, beta and diversity range) of plants for each treatment (A, C), as well as the comparison between the results observed and the results expected by random (B, D) of riparian forest in the River Piranga basin, Brazil. The dashed line in B and D represents the observed values and the columns the simulated values expected by random.

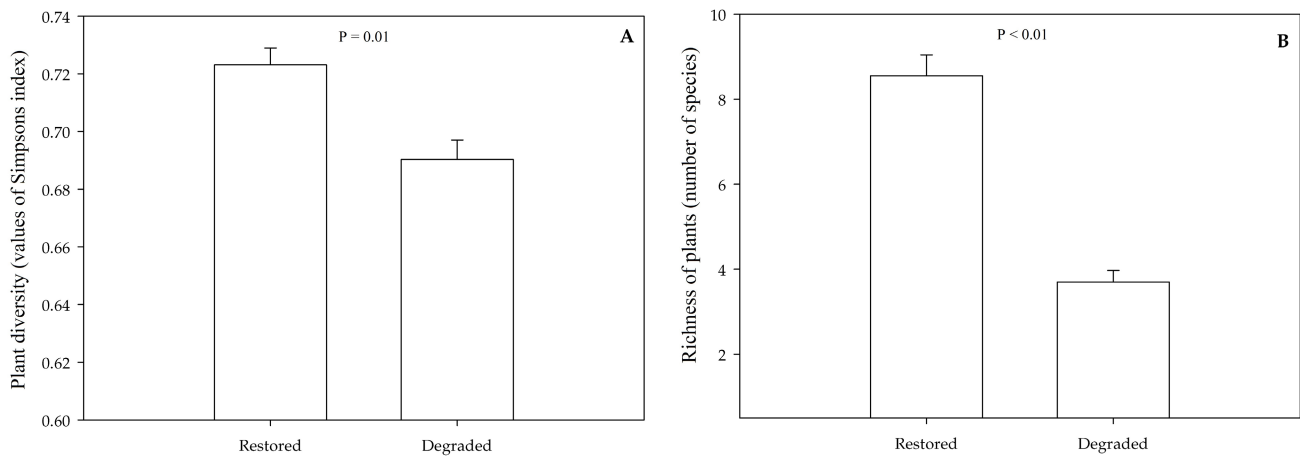


Fig. 3. Diversity (A) and richness of species (B) of riparian forest in the River Piranga basin (Brazil) for Restored and Degraded treatments. The columns indicate the average values and the bars show standard errors.

Table 2. Summary of statistical results for models tested. The rows separate each constructed model

Models		Df	AIC	BIC	logLik	Deviance	Chisq	Chi Df	p-value	SD (Random effects)
Simpson diversity × Environment	Null	3	-161.89	-156.82	83.94	-167.89	5.8594	1	0.01549	0.02943
	Alternative	4	-165.75	-158.99	86.87	-173.75				
Richness × Environment	Null	2	170.22	173.59	-83.11	166.22	7.9304	1	0.004861	0
	Alternative	3	164.29	169.35	-79.143	158.29				
Alpha diversity × Environment	Null	3	265.74	282.55	-129.87	259.74	23.377	1	1.33 × 10⁻⁶	3.2240
	Alternative	4	244.37	266.77	-118.18	236.37				
Gamma diversity × Environment	Null	3	8690.60	8707.40	-4342.30	8684.60	18.723	1	1.51 × 10⁻⁵	10.929
	Alternative	4	8673.90	8696.30	-4332.90	8665.90				
Gamma diversity × Environment	Null	3	5092.20	5109.00	-2543.10	5086.20	16.9	1	3.94 × 10⁻⁵	1.1275
	Alternative	4	5077.30	5099.70	-2534.70	5069.30				
Native richness × Environment	Null	2	369.28	376.26	-182.64	365.28	6.3092	1	0.01201	0
	Alternative	3	364.97	375.44	-179.49	358.97				
Ruderal richness × Environment	Null	2	405.44	412.41	-200.72	401.44	5.492	1	0.0191	0
	Alternative	3	401.94	412.41	-197.97	395.94				
Alien richness × Environment	Null	2	167.50	174.48	-81.75	163.50	0	1	1.00	0.2665
	Alternative	3	382.28	392.74	-188.14	376.28				
Abundance × Status:Environment	Null	2	2254.10	2261.10	-1125.00	2250.10	115.42	5	2.20 10⁻¹⁶	0
	Alternative	7	2148.70	2173.10	-1067.30	2134.70				

Note: Df – degrees of freedom; AIC – Akaike information criterion; BIC – Bayesian inference criteria; logLik – log-likelihood; Chisq – Chi-square; Chi Df – Chi-square degrees of freedom; SD – standard deviance. The p-values < 0.05 are in bold.

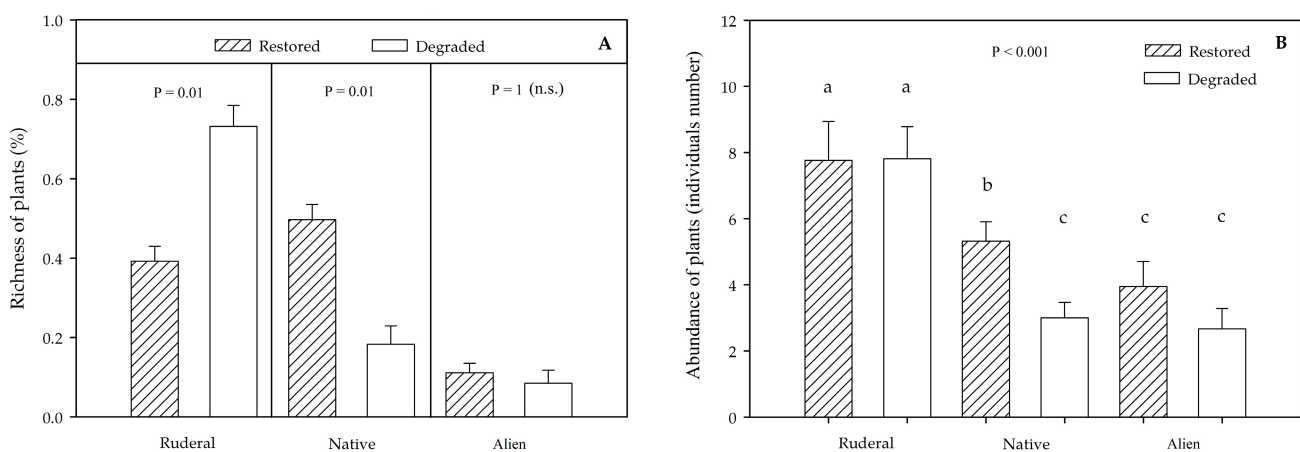


Fig. 4. Richness (A) and abundance (B) of riparian forest in the River Piranga basin (Brazil) for Restored and Degraded treatments for each class, namely native, ruderal, and alien. Columns indicate the average values, and bars show standard errors. In A, «n.s.» – Non significant; in B, the statistical equal means are shown by equal letters above the bars.

Overall the abundance of the native plants was almost twice higher ($\bar{x} = 6.53$, $SE = \pm 0.49$) than of alien plants ($\bar{x} = 3.64$, $SE = \pm 0.59$) ($F_{(1, 239)} = 6.7462$, $p = 0.009$). On the other hand, when the ruderal natives are considered, the abundance of plants varied according to the interaction between environment and classification of species, as well as among the classification status of the species, but did not differ when compared only between environments (Table 2). Thus, the colonisation by ruderal plants was higher at the expense of the native and alien plants, independently of the environment (Fig. 4B). Native plants in the Restored environment were more abundant than the alien plants (Fig. 4B). In contrast, in the Degraded environment, native and alien plants showed similar abundance (Fig. 4B).

Discussion

The plant diversity within treatment is higher on the Restored site (by using nurse plants) than of Degraded treatment (by pasture abandonment). This was verified as the observed values of the diversity partition (α , γ , and β) were higher in the Restored environment. Richness, alpha- and gamma-diversity index indicate that the introduction of *B. dracunculifolia* in the Degraded areas facilitated plant species enrichment in those treatment. Similar results for richness were observed along roads of montane Neotropics (Perea et al., 2019), other congeneric species, such as *Baccharis conferta* Kunth (Sánchez-Velásquez et al., 2011) and *B. uncinella* DC. (Duarte et al., 2006), have been argued to be facilitators, the effect was described for certain target species (*Albies religiosa* (Kunth) Schldl. & Cham. and *Araucaria angustifolia* (Bertol.) Kuntze, respectively) and not for the natural recovery of species diversity. However, the prediction that «plant diversity would be higher on the Restored site, in the presence of *B. dracunculifolia*, at the expense of the Degraded site» was not corroborated. Although the diversity index, richness as well as alpha and gamma values of the simulated communities were greater in the Restored site, the beta-diversity was larger in the Degraded site.

It is important to note that, although the species composition (beta-diversity) on simulated communities were larger in the Degraded environment, we noticed a higher number of ruderal and low number of native species in the Degraded site. In the context of land recovery, this finding is not desirable in a degraded environment,

because it indicates that the disturbance remains (Bailey et al., 1998; Brockerhoff et al., 2003; González et al., 2016).

Although unexpected, the beta diversity in modified environments may increase due to the invasion of these habitats by ruderal or alien species. The increase in the beta-diversity of plants was already described for landscapes with high deforestation (Arroyo-Rodríguez et al., 2013), in habitats disturbed by fire (Myers et al., 2015) or in eroded regions afforested with invasive species (Kou et al., 2016). Socolar et al. (2016) explain that this increase in beta-diversity in modified habitats can happen at the first moment. However, the beta-diversity decreases over time or when the disturbance persists because only a few species resistant to the disturbance remain in the habitat. Over time, there will be a homogenisation of the plant community occurring in that habitat followed by a probable dominance of species with invasive characteristics.

Overall, the restoration through the planting of *B. dracunculifolia* increased about twice the plant species richness in the Restored environment. It is worth mentioning that the abundance of native plants in the Restored environment was higher than in the Degraded environment. In this study, 17 unique native plants were found in the Restored environment while the frequency of ruderal species was recorded lower (47% in the Restored environment, and 88% in the Degraded environment). This indicates that the planting of the nurse species may have influenced the environmental conditions of the Restored area, favouring native species. Castro et al. (2002) demonstrated how the planting of nurse species softens the environmental harshness, filtering the solar radiation through the canopy shade. These environmental changes attributed to the nurse plants are responsible for creating microclimates that can favour the arrival of native species originated from the neighbouring forest remnants (Filazzola & Lortie, 2014). We argue that the combination of these factors may have contributed to the differences found in the structure of the studied plant communities. On the other hand, there are reports of species-specific negative effects between ontogeny of nurse species and the development of target species (Paterno et al., 2016; Fagundes et al., 2018). Therefore, future studies need to focus on the understanding of which environmental parameters and which traits of each species would be

improved under the influence of the nurse species *B. dracunculifolia*.

On the other hand, the predictions that «the abundance of native plants is higher than aliens in the Restored environment» and «the abundance of native plants is higher than ruderal plants in the Restored environment» were not corroborated. Considering that the classification of ruderal plants is intrinsically related to the beginning of a colonisation process (Trnkova et al., 2010), long-term sampling would be important to capture the possible variation of this classification between environments. This study indicates that the colonisation by species of later successional stages in the Restored area is already in progress. An evidence of this process is the high frequency of the late succession species, *Piptadenia communis* Benth., whose occurrence was exclusive to the Restored areas. This species is also distributed in the forests surrounding the Degraded area but did not colonise it. The fact that the abundance of ruderal plants does not vary between environments reinforces the idea that the process of colonisation of the Restored area is still in the early stages. An example of this is that the most abundant species in both environments of this study is a ruderal pioneer *Sida cordifolia* L. This species was found at a frequency of 48% in the Degraded environment and 25% in the Restored environment. Importantly, the Restored area showed a lower alien plant diversity as compared to the Degraded area, which agrees with previous studies arguing that *Baccharis dracunculifolia* contributes to controlling alien plant invasion in disturbed treatment (Perea et al., 2019).

Conclusions

We conclude that *B. dracunculifolia* planting played an important role in increasing native plant diversity and reducing both ruderal and alien richness in previously disturbed areas. *Baccharis dracunculifolia* is widely distributed in Brazil through Cerrado and Atlantic Rainforest biomes. Therefore, such observations are of great relevance in the use of this species in ecological restoration projects. In addition, this study is a pioneer in documenting the facilitator effect of *B. dracunculifolia* in areas of gallery forest of the Atlantic Rainforest degraded for mining.

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РОЛЬ КУСТАРНИКОВ *VACCHARIS* (ASTERACEAE) В КРАТКОВРЕМЕННОМ ВОССТАНОВЛЕНИИ АТЛАНТИЧЕСКОГО ЛЕСА

У. К. Сикейра¹ , А. Л. Моура² , Р. Переа³ , М. Дж. П. Алвес⁴ , Ж. В. Фернандес¹ 

¹Федеральный университет Минас-Жерайса, Бразилия
e-mail: kenedy.siqueira@gmail.com, gw.fernandes@gmail.com

²Обернский университет, США
e-mail: arthurlatouniertoura@gmail.com

³Мадридский политехнический университет, Испания
e-mail: ramon.perea@upm.es

⁴Федеральный университет штата Пара, Бразилия
e-mail: michelijacobyperreiraalves@gmail.com

Интродукция видов-нянь на нарушенных участках постоянных особо охраняемых природных территорий представляет собой полезную стратегию восстановления растительности в прибрежных лесах. Виды рода *Vaccharis* (Asteraceae) были задокументированы как потенциальные растения-няни, способные реструктурировать местное растительное сообщество. Это исследование было направлено на оценку потенциальной роли растения-няни для *Vaccharis dracunculifolia* в кратковременном восстановлении местного растительного сообщества нарушенного атлантического тропического леса. Исследование проводилось на двух заброшенных пастбищах в прибрежном лесу в бассейне реки Пиранга. На одном из них была посажена *V. dracunculifolia* (восстановленный участок) для вклада в восстановление растительности. На втором участке не предпринимались никакие вмешательства (деградированный = контрольный участок). Отбор проб проводился через 18 месяцев после посадки *V. dracunculifolia*. Закладывали по 20 площадок 2×2 м в каждом варианте эксперимента ($n = 40$ площадок), на которых регистрировали все виды растений (за исключением транслоцированных особей *Vaccharis* и злаков), классифицируя их как аборигенные, рудеральные или чужеземные. Видовое богатство растений было в два раза выше на восстановленном участке, чем на деградированном участке. Кроме того, наблюдаемые значения альфа-, гамма- и бета-разнообразия также были выше при восстановлении. На восстановленном участке было отмечено 17 эксклюзивных (встреченных только там) аборигенных видов, в то время как на деградированном участке было только три не эксклюзивных аборигенных вида. Кроме того, на восстановленных участках с *V. dracunculifolia* отмечено меньшее количество рудеральных и чужеземных видов по сравнению с деградированными участками. Мы пришли к выводу, что даже в короткие сроки посадка *V. dracunculifolia* оказала положительное влияние на формирование местного растительного сообщества и возможное снижение вероятности инвазии чужеземными видами.

Ключевые слова: бассейн реки Пиранга, инвазия растений, колонизация видов, нулевая модель, поддержка растений, составное сообщество, экологическое восстановление