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## Functional diversity of herbaceous species under different fire frequencies in Brazilian savannas

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### ABSTRACT

Fire plays an important role in determining the structure of savannas, so that frequent fires tend to favor the herbaceous species in savannas. Functional diversity is a measure of the range of the species functional traits and is thought to be related to many functions and properties like community stability, nutrient cycling, and productivity, for example. We examined if functional diversity of herbaceous species was different along a fire gradient, and if observed functional diversity differed from a random assembly of species traits. We sampled three sites with different fire frequencies. According to the intermediate disturbance hypothesis, we expected the functional diversity to be higher in the intermediate fire frequency site. Subsequently, we excluded rare species from the analyses – species with less than ten individuals in each site – to verify their contribution to the observed patterns of functional diversity. We expected to find no difference with the exclusion of rare species. Functional diversity values, either considering all species or excluding the rare ones, were not different among sites, suggesting that (1) frequent fires may not affect overall functioning of the herbaceous community in savannas and (2) rare herbaceous species seem to present lower functional complementarity. According to the neutral theory, most of the herbaceous species may have evolved stochastically in niches more similar than competitive exclusion could allow. Also, functional diversity was not different from what one may expect by chance in any of the three sites. We also calculated the floristic similarity among sites. When we considered qualitative data, we found floristic differences between the sites of high and intermediate as well as between high and low fire frequency sites, but there was no difference between the intermediate and low fire frequency situation. Floristic similarities were different among all the three sites when the abundances of species were considered. Consequently, different fire conditions may change species composition without affecting the functional diversity of the herbaceous layer of savannas, corroborating the idea that savannas are more stable in functional than in floristic terms.

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### Introduction

Fire influences the structures of savannas all over the world through complex effects, such as changes in light availability, soil surface temperature, plant nutrient availability, and the amount of accumulated dry biomass on the ground (Bond et al., 2005; Cook, 2003; Miranda et al., 1993; Nardoto et al., 2006). Since savanna fires are surface fires, it is the herbaceous layer that is mainly burned (Andersen, 2003; Miranda et al., 1993). Recurring fires are expected to have an important effect on the diversity of the ground layer plant communities (Coutinho, 1990; Ramos-Neto and Pivello, 2000).

In South America, the largest savanna region is the Brazilian cerrado (Gottsberger and Silberbauer-Gottsberger, 2006). Neotropical savannas burn generally at intervals of 1–3 years (Coutinho, 1990; Moreira, 2000) and, fire exclusion leads to an increase in the herbaceous standing biomass (Govender et al., 2006). The cerrado, as other savannas, presents an herbaceous and a woody layer, whose relative predominance is controlled considerably by fire regimes (Coutinho, 1990; Hoffmann, 1998; Moreira, 2000). Savannas are habitats that have experienced fire for 20 million years at least (Bond et al., 2003), and herbaceous species are thought to be favored by this disturbance (Coutinho, 1990; Moreira, 2000). Frequent burnings tend to favor herbaceous species due to their protected belowground meristems (Gottsberger and Silberbauer-Gottsberger, 2006; Moreira, 2000). The reproductive output of many herbaceous species is increased by recurrent fires (Sarmiento, 1992), and the persistence of some species in the community depends on high fire frequency (Canales et al., 1994).

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Functional diversity is a measure of the range of the species' functional traits (Petchey and Gaston, 2002) that are important in determining functioning of a plant community (Díaz and Cabido, 2001; Tilman et al., 2001). The processes related to community functioning seem to be better understood when assessed by functional traits rather than by the number of species per se (Díaz and Cabido, 2001). For example, Fornara and Tilman (2008) found that trait complementarity increases soil carbon and nitrogen accumulation, being positively associated with root biomass. Communities with higher functional diversity are also expected to have higher productivity (Tilman et al., 1997), resilience (Nystrom and Folke, 2001), and better resistance to plant invaders (Dukes, 2001; Prieu-Richard and Lavorel, 2000). In addition, trait distributions can be used to test hypotheses on how local communities are assembled.

The intermediate disturbance hypothesis predicts that diversity is higher when disturbances levels are neither too high nor too low (Connell, 1978). According to this hypothesis, we may expect the functional diversity to be higher in a site with intermediate fire frequency, because high or low fire frequencies tend to narrow the trait differences among species, selecting those with similar traits (Chase, 2003; Díaz et al., 1998). Conversely, species with different set of traits tend to occur when fire frequency is intermediate, because, under intermediate levels of disturbance, species of both limits of the fire gradient may co-occur (Connell, 1978), maximizing the functional diversity.

Although there are some studies of fire effects on savanna trees (see Moreira, 2000, for references), responses of herbaceous species to different fire regimes are still poorly understood (Andersen, 2003). Understanding how vegetation responds to different fire frequencies is fundamental for a successful fire management (Andersen et al., 2003). Here, we examined the functional diversity of the herbaceous layer in cerrado sites under different fire frequencies. Since most species tend to occur in relatively low abundance in natural communities (Gaston, 1994; Magurran, 2004), it is usually assumed that rare species will have low impact on community functioning, being analogues of the dominant species (Cornelissen et al., 2003; Grime, 1998). However, there is evidence that rare species may play an important role in community functioning (see Lyons et al., 2005 for a review). So, we also calculated functional diversity with and without considering rare species to explore their role in cerrado plant communities. If common species were more important to community functioning than rare ones, we expected to find no difference in the results when excluding rare species from the analyses. In addition, we investigated the floristic similarity among the study sites and how the common and rare species composition differed among them.

In a nutshell, we addressed the following questions: (1) Are there differences in functional diversity among the cerrado sites with different fire frequencies? (2) Does functional diversity differ in each site from that expected by the chance assembly of communities? (3) Does the exclusion of rare species from the analyses alter the observed patterns? (4) Are differences in floristic composition followed by differences in functional diversity?

## Materials and methods

### Study site

We collected our data in Emas National Park (ENP), located in the Brazilian Central Plateau, southwestern Goiás State (17°49'–18°28'S; 52°39'–53°10'W) in 2006. The ENP comprises 132,941 ha and is one of the largest and most important reserves in the cerrado region (UNESCO, 2001). According to Köppen's (1931) system, regional climate is classified as Aw, a humid tropical climate, with wet summer and dry winter. Annual rainfall and mean

temperature lie around 1745 mm and 24.6°C, respectively. In the park, there is a gradient from open to closed savannas, and also wet grasslands, riparian forests, and semideciduous forests (Ramos-Neto and Pivello, 2000).

Up to 1984, the ENP was used by farmers for cattle ranching, and forage regrowth was promoted every year in dry season. After that year, the reserve was fenced and a fire policy was established (Ramos-Neto and Pivello, 2000). Uncontrolled wildfires used to occur every 3–4 years burning 80% of the park, on average (França et al., 2007; Ramos-Neto and Pivello, 2000). Since 1994, preventive firebreaks are burned annually during the dry season, an area of approximately 10 km<sup>2</sup>. A fire brigade stays in the park to prevent fire during the dry season (França et al., 2007). As a consequence, there are currently few occurrences of anthropogenic fires inside the park. The actual proportion burned each year is less than 30% of the total area, and the frequency of recurring fire at a given point is 6–7 years on average (França et al., 2007).

### Ecological data

We surveyed three nearby sites with different fire frequencies, but with similar physiognomies (grassland cerrado), on the same soils (Oxisols), in the southeastern portion of the ENP. The first site (approximately 18°18'50'S and 52°54'00'W) was a firebreak, burned annually since 1994. The second site (approximately, 18°19'01'S and 52°54'10'W) was also a firebreak, but burned approximately every 2 years since 1994. The third site (approximately, 18°17'28'S and 52°53'41'W) was a protected cerrado, unburned since 1994. In physiognomies similar to those studied here, biomass of herbaceous undershrub species normally ranges from 350 to 550 g m<sup>-2</sup> after 1–3 years since last fire (Kauffman et al., 1994; Sarmiento and Vera, 1979). The herbaceous undershrub species are the principal fuel load in savannas, presenting a combustion factor higher than 90% (Kauffman et al., 1994). So, fire exclusion for more than a decade (as it is the case for the unburned site studied here) is a matter of concern because it leads to a higher amount of standing alive and dry biomass which may result in uncontrolled fires.

In each site, we placed a 1250 m line, with 125 points, 10 m apart one from the other. In each point, we sampled four individuals in the herbaceous layer with the point-quarter method (Müller-Dombois and Ellenberg, 1974), summing up 500 individuals in each site. We defined the herbaceous layer as all herbaceous individuals and all woody individuals with stem diameter at soil level lower than 3 cm. We did not sample seedlings, defined as those individuals with cotyledons. We identified the individuals by comparing the collected material to reference collections from ENP (Batalha and Martins, 2002) and vouchers lodged at the State University of Campinas herbarium. When identification at species level was not possible, we classified these plants as morphospecies.

To estimate functional diversity, we selected five traits related to fire (Cornelissen et al., 2003; Müller et al., 2007): basal area, plant height, specific leaf area, dispersal mode, and growth form. We sampled 10 individuals of each species to calculate the average for each functional trait. When there were not ten individuals of a given species, we used all individuals sampled. Basal area is a trait related to space occupation, resistance to competition, and resource acquisition, which all are important after fire events (Navas and Moreau-Richard, 2005). Plant height is associated to competitive ability, fecundity, and the interval between two disturbances, such as burnings (Cornelissen et al., 2003). Plants submitted to disturbances like fire may invest on resistant leaves, which will present lower values of specific leaf area (Cornelissen et al., 2003; Westoby, 1999). We measured leaf area with the ImageJ software (Rasband, 2004). Dispersal mode is a categorical measure, related to colonizing abilities and disturbances (Pausas and Paula, 2005). Growth

**Table 1**  
Observed functional diversity (FD) between three herbaceous communities under different fire frequencies in Emas National Park, central Brazil (17°49'–18°28'S and 52°39'–53°10'W) and the corresponding lower and upper tails from 10,000 simulated functional diversities ( $D_{sim}$ ). HiFi: high fire frequency; InFi: intermediate fire frequency; LoFi: low fire frequency.

Functional diversity approach	Observed FD values		$D_{sim}$ distribution	
			Lower tail (2.5%)	Upper tail (97.5%)
All species included	HiF	0.51	0.41	0.61
	InFi	0.41	0.37	0.57
	LoFi	0.58	0.43	0.63
Rare species excluded	HiFi	0.12	0.07	0.16
	InFi	0.11	0.08	0.17
	LoFi	0.17	0.10	0.19

form is also a categorical measure, mainly determined by above-ground structure and height, and is associated with the general ecological strategy of plants (Cornelissen et al., 2003). We measured these traits according to Cornelissen et al. (2003).

### Functional diversity

We used the functional diversity index (FD) proposed by Petchey and Gaston (2002, 2006) to measure the functional diversity of the sites. FD is the total branch length of a functional dendrogram, measuring the extent of complementarity among species trait values by estimating the species dispersion in trait space (Petchey and Gaston, 2002, 2006). The larger the differences among species trait values, the larger the complementarity and, thus, the higher the functional diversity. First, we assembled a trait matrix, with species in rows and functional traits (average values) in columns. We standardised the trait values to zero mean and unit variance. Then, we used the Gower distance to convert the trait matrix into a distance matrix, from which we produced a dendrogram by using the unweighted pair group method with arithmetic mean clustering algorithm. Finally, we calculated the total branch length of the dendrogram necessary to connect all species within each site that determined the FD values. We standardised FD to vary from zero to one.

For comparisons of functional diversities, we applied a randomisation test (Manly, 1997), with 10,000 permutations, to check whether FD values of each site were significantly different ( $\alpha=0.05$ ). First, we calculated the difference  $D_{1-2}$  between the FD values of any two sites ( $D_{1-2} = |\text{FD}_{\text{site 1}} - \text{FD}_{\text{site 2}}|$ ). Then, we randomly re-sampled the species of both sites and determined the difference  $D_{sim}$  between the simulated FD values obtained – a step repeated 10,000 times, which provided us with a distribution of  $D_{sim}$  values. Then, to obtain a  $P$ -value, we calculated the proportion of all  $D_{sim}$  values that were higher than or equal to  $D_{1-2}$ .

To answer whether functional diversity differed from what can be expected by chance assembly of communities, we constructed a null-model considering that all species sampled in the three sites had the same chance to occur in one of those sites. First, we randomised the species composition (including all species observed in the three sites), then we created simulated assemblages without replacement, keeping the observed species richness for each site, and finally we calculated a simulated FD for each site. We repeated this procedure 10,000 times to generate a distribution of simulated FD values for each site and then estimated whether the observed FD values were within the empirical 95% confidence intervals of the simulated distribution. So, we could test how different fire frequencies would assemble the species in respect of their functional attributes. We conducted all analysis in R (R Development Core Team, 2008).

We computed the FD of each site considering all species sampled in the first analysis and, then, excluding rare species. Functional

information was site-specific, and we considered the species sampled in each site separately, allowing us to include intraspecific differences among sites (Cianciaruso et al., 2009). We considered as rare those species with less than ten individuals in each site. Although other studies have usually included only the most abundant species (see Cornelissen et al., 2003), here we also included information on rare ones. If we did not consider rare species, we would have excluded most of the species sampled and, consequently, would have lost the generalization power of our analyses (Vepsäläinen and Spence, 2000; Westoby, 1999). In this approach, overall differences among the studied sites are more important than the exact magnitudes of species traits (Vepsäläinen and Spence, 2000).

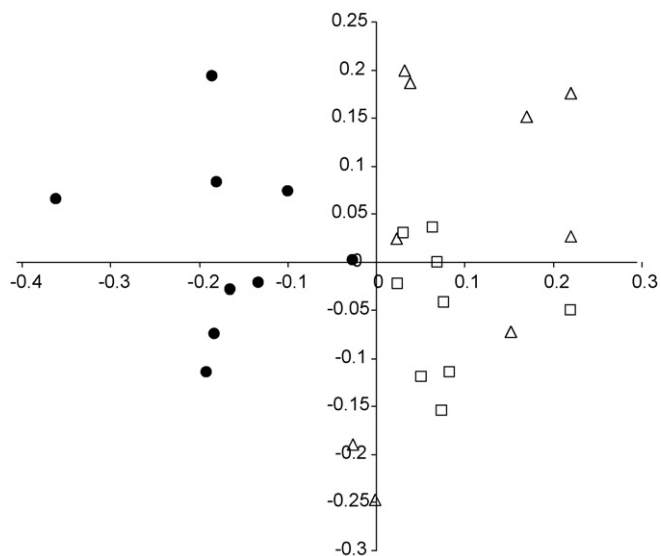
### Floristic composition

In each site, we merged the sampling points in nine groups, that is, from points 1 to 12, 13 to 24, 25 to 36, and so on. So, for each site, we had nine groups, eight with 56 individuals and one with 52 individuals. This merging was necessary to allow floristic comparisons among sites. Then, we constructed a matrix, based on which we calculated floristic similarity among a group with Bray–Curtis index, both quali- and quantitatively (Magurran, 2004). Finally, to test whether the three sites differed in relation to their floristic composition ( $\alpha=0.05$ ), we did a non-parametric multivariate analysis of variance (Anderson, 2001) and a pair-wise a posteriori test, with 10,000 permutations. We repeated this procedure, excluding the rare species. Also, we did a non-metric multidimensional scaling (NMDS) to visualize data dispersion in multidimensional space (Clarke, 1993; Faith et al., 1987).

### Results

We sampled 97 species in the three sites, 57 (6 common and 51 rare) species in the high fire frequency site, 50 (7 common and 43 rare) species in the site with intermediate fire frequency, and 61 (9 common and 52 rare) species in low fire frequency site (see Appendix A). We excluded from the analyses four species in the high fire frequency site (comprising 7% of all species sampled and 1.6% of the individuals sampled), two species in the intermediate fire frequency site (comprising 4% of all species sampled and 0.4% of the individuals sampled), and one species in low fire frequency site (equivalent to 1.6% of all species sampled and 0.2% of the individuals sampled) because we were unable to estimate their leaf area with precision.

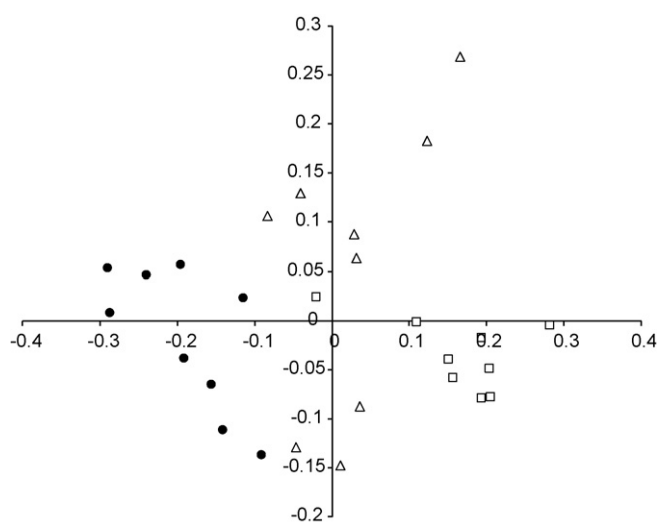
FD values in the sites with high, intermediate, and low fire frequencies were, respectively, 0.51, 0.41, and 0.58. These values were not significantly different from each other ( $P>0.05$ ). Functional diversity was not different from what one may expect to occur by chance in any of the three sites (Table 1). When we excluded rare species from the analyses, values of FD were 0.12 in the high fire frequency, 0.11 in the intermediate fire frequency, and 0.17 in the



**Fig. 1.** Non-metric multidimensional scaling of qualitative (presence–absence) floristic similarities with all species included. We find differences between high and intermediate fire frequency sites ( $P=0.001$ ) and between the high fire frequency and low fire frequency sites ( $P=0.001$ ), but no difference between the intermediate fire frequency and low fire frequency sites ( $P=0.07$ ). ● = high fire frequency site, □ = intermediate fire frequency site, Δ = low fire frequency site.

low fire frequency ( $P>0.05$ ). Once again, there were no significant differences between the values (Table 1).

Using qualitative data (presence/absence) we found a significant dissimilarity in the floristic composition between the high and intermediate fire frequency situations and between high and low fire frequency situations, but there was no difference between intermediate and low fire frequencies (Fig. 1). However, when we took into account species abundance all sites were different among each other (Fig. 2). Exclusion of rare species from the analysis produced the same patterns; therefore we just present here the results of NMDS to all species (Fig. 2).



**Fig. 2.** Non-metric multidimensional scaling of quantitative (including species abundances) floristic similarities with all species included. We find differences between high and intermediate fire frequency sites ( $P=0.0001$ ) and between the high and low fire frequency sites ( $P=0.0001$ ), but no difference between the biennially and the unburned sites ( $P=0.0005$ ). ● = high fire frequency site, □ = intermediate fire frequency site, Δ = low fire frequency site.

## Discussion

Our initial expectation was that functional diversity would be higher under intermediate fire frequency, corroborating the intermediate disturbance hypothesis (Connell, 1978). We also expected that high fire frequency would restrict the range of functional traits and, thus, reduce functional diversity. Nonetheless, fire did not change the functional diversity of the herbaceous species of cerrado.

Differently to traditional competitive-based explanations (Hutchinson, 1959; Leibold, 1998), the neutral theory of biodiversity (Hubbell, 2001) may explain our results. According to this theory, converging species should coexist for arbitrarily long periods and not be subjected to competitive exclusion by other species (Hubbell, 2006). As a result, a great number of species could evolve adaptations for the most frequent environmental conditions (Hubbell, 2006). Thus, it is quite likely that herbaceous species of the cerrado have evolved stochastically niches more similar than allowed by competitive exclusion (see Hubbell, 2006, for forest species), taking into account that the cerrado flora has been shaped for millions of years (Bond et al., 2003; Gottsberger and Silberbauer-Gottsberger, 2006) to resist or even to depend on fire. Considering this, different fire conditions may change species composition without affecting the functional diversity of the herbaceous layer in cerrado.

Concerning floristic similarity, the high fire frequency site was different from the others, if only species presences are considered. But, if species abundances are taken into account all sites were different. Protection against fire usually brings a gradual and progressive change in the flora and in the entire vegetation (Coutinho, 1990). In the cerrado, the majority of species are highly resistant to fire, but present different strategies. Most species are perennials exhibiting subterranean organs such as bulbs, underground shoots, rhizomes, and xylopodia which, owing to the soil insulation, escape the destructive action of fire (Coutinho, 1990; Moreira, 2000). The buds of tunic-graminoids are protected inside the densely imbricated sheaths of their leaves where combustion is limited due to inadequate aeration (Gottsberger and Silberbauer-Gottsberger, 2006). Yet, there are species that although woody, belong to the herbaceous undershrub stratum, developing their entire system of trunks and woody branches subterraneously, and only the small vegetative branches or yearly reproductive sprouts appear above the soil. So, given these different strategies one would expect that changes in species composition should be reflected in differences of functional diversity. However, this was not the case. High fire frequencies obviously promoted a floristic replacement without significant changes in functional diversity. This corroborates the idea that savannas are more stable in functional than in floristic terms (Sarmiento, 1996). This has the implication that communities with different species composition do occupy a similar portion of the multidimensional trait space, that is, even if species identity is different the niche space they are filling is similar.

However, the measure of functional diversity we used does not account for species abundance. Indeed, there is an intuitive reason for including abundances on functional diversity (Mouillot et al., 2005; Mason et al., 2005), but up to now all measures of functional diversity that include abundances have shortcomings (see Petchey and Gaston, 2006). When species abundance is included, the observed patterns of floristic similarity among sites change. Thus, abundance may be relevant as a component of functional diversity as well. But, how to include this properly into the calculations is yet an open question.

Fire frequency in the ENP was high in historical times (França et al., 2007) and may have selected plant species with similar functional traits that confer advantages to individual survival. Before establishment of a fire management, in 1994, uncontrolled fires

occurred every 3–4 years (França et al., 2007; Ramos-Neto and Pivello, 2000). After the establishment of firebreaks to prevent this high frequency and intensity of burning, fire frequency at a given point became reduced to 6–7 years on average. The high fire frequency may have reduced the species of the regional pool to those species that are fire tolerant and, consequently, present similar functional traits. As a result, species assemblages of sites under actually different fire frequencies tend to present similar functional diversities. Therefore, the similarity in functional diversity we found may be also a consequence of the historical trait selection for fire resistance in ENP species pool.

When we excluded rare species from the analyses – species with less than ten individuals per site, we also did not find significant differences in functional diversity. In other communities, rare species were identified as keystone species in soil resource dynamics, making substantial contributions to community processes (Lyons et al., 2005). Furthermore, around 80% of the total functional diversity of the herbaceous layer of such sites was due to differences in the traits of the rare species showing that these species are very complementary among each other. This may indicate that – although the effect of rare species on functional diversity did not alter the pattern found in different fire frequencies – they increase the functional diversity measure being complementary to each other.

Considering that differences in annual fire regimes are not mirrored in the functional diversity of the herbaceous communities, other fire oriented management strategies (for example, patch burnings; Ramos-Neto and Pivello, 2000) might be also a good policy of conservation of the cerrado. It is recognized that these strategies control more efficiently biomass accumulation and reduce consequently the risk of uncontrollable wildfires (Govender et al., 2006). Such management has been used for decades in other savannas around the world (Govender et al., 2006), but up to now it has not been implemented in Brazilian savannas, even if some stud-

ies have this recommended (Pivello and Norton, 1996; Ramos-Neto and Pivello, 2000). This is mainly due to both a lack of knowledge about the impact of fires on biodiversity and a prejudice that fire is always harmful. Our results provided additional evidence that it is possible to have a more diversified fire management by introducing planned fires without losing biodiversity.

In conclusion, we showed that the herbaceous layer of the cerrado is functionally resistant to different and also high fire frequencies. The frequency of fire alters the floristic composition, revealing that some species are restricted to some fire frequencies, but does not change the functional diversity. Thus, the cerrado vegetation seems to be more stable in functional than in floristic terms. To maintain high levels of biodiversity in the communities studied, it seems to be important to promote different fire frequencies: This will allow the presence of different species under different fire frequencies without changing the overall functional diversity of the communities.

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### Appendix A.

Table A.1.

**Table A.1**

Species sampled in the ground layer of savannas under different fire frequencies in Emas National Park, central Brazil (17°49'–18°28'S and 52°39'–53°10'W). Average values are shown for continuous traits: basal area (BA, cm), plant height (H, cm), and specific leaf area (SLA, mm<sup>2</sup> mg<sup>-1</sup>). N.ind: number of individuals sampled. For categorical traits: disp: dispersal mode classified as anemo-, auto-, epizoo-, endozoo-, and synzoochory; gr: growth form classified as long basal (lb), dwarf shrub (ds), tussock (t), tree (tr), climber (cl), cushion (cs), and erect leafy (el).

Species	High fire frequency			Intermediate fire frequency			Low fire frequency			disp.	gr.			
	BA	H	SLA	N.ind	BA	H	SLA	N.ind	BA			H	SLA	N.ind
<i>Allagoptera campestris</i> (Mart.) Kuntze	2.87	33	59.54	1	0.62	34.4	60.39	11	1.59	28	59.54	1	endo	ds
<i>Anacardium humile</i> A. St.-Hil.	1.21	21.08	51.89	15					0.41	21.56	70.2	11	endo	ds
<i>Andira laurifolia</i> Benth.									0.11	2	71.57	1	endo	ds
<i>Andropogon leucostachyus</i> Kunth	1	33	129.26	1	0.25	29	77.77	1					anemo	t
<i>Anemopaegma acutifolium</i> DC.					0.21	18	57.4	1					anemo	ds
<i>Anemopaegma arvense</i> (Vell.) Steilfeld & J.F. Souza	0.4	8	116.76	6	0.2	9	99	3	0.21	14.7	108.48	5	endo	ds
<i>Aspilia leucoglossa</i> Malme	1.71	33.38	49.98	13	2.98	70	68.38	2					epi	el
<i>Axonopus barbigerus</i> Hitchc.	3.65	57.5	139.85	3	0.95	40.45	77.03	102	2.31	37.11	96.41	37	auto	t
<i>Axonopus pressus</i> (Nees ex Steud.) Parodi	3.71	34.43	61.25	7	0.93	33.2	63.18	88	1.82	24.58	68.57	27	auto	t
<i>Axonopus suffultus</i> (Mikan ex Trin.) Parodi	1.94	25.8	108.42	40									anemo	s
<i>Banisteriopsis campestris</i> (A. Juss.) Little	2.54	61	71.4	1	0.33	4	281.33	1	1.5	140	73.55	1	auto	tr
<i>Bauhinia rufa</i> (Bong.) Steud.									0.83	7.8	254.78	5	auto	t
<i>Bulbostylis junceiformis</i> (Kunth) C.B. Clarke	1.83	4.5	175.77	9					0.56	17	57.43	1	endo	s
<i>Byrsionina guillemianiana</i> A. Juss.	0.49	13.3	52.47	7	0.2	20.33	66.64	3	0.57	41.8	65.27	5	endo	s
<i>Campomanesia adamantium</i> (Cambess.) O.Berg	0.3	19.5	60.51	4	0.22	22	387.41	3	0.38	26.5	55.23	2	endo	s
<i>Campomanesia pubescens</i> (Mart. ex DC.) O.Berg	0.23	11.88	74	4	0.12	6	140.64	1	0.24	5.25	125.12	2	endo	ds
<i>Casearia</i> sp. Jacq.	0.1	8	104.85	1					0.3	39	76.33	1	endo	t
<i>Casearia sylvestris</i> Sw.									0.1	10	97.64	1	endo	cl
<i>Chayaponia espelma</i> (Silva Manso) Cogn.													auto	s
<i>Chamaecrista setosa</i> (Vogel) H.S. Irwin & Barneby	0.1	16	70.9	2					0.41	25	49.69	3	endo	s
<i>Chromolaena squallida</i> (DC.) R.M. King & H. Rob.	0.78	56.75	104.43	8	0.12	12	58.22	1	0.16	20	159.34	1	syn	ds
<i>Cordia sessilis</i> (Vell.) Kuntze									0.14	3	111.11	1	syn	ds
<i>Croton glandulosus</i> L.													syn	ds
<i>Croton grandivolum</i> Bail.					0.22	18	191.38	2	3.01	34.14	151.78	1	epi	t
<i>Croton</i> sp. L.													endo	ds
<i>Ctenium chapadense</i> Trin.					0.24	32.5	80.59	2	0.11	6.75	131.41	2	auto	ds
<i>Declieuxia fruticosa</i> (Willd. ex Roem. & Schult.) Kuntze	0.2	26	85.78	1	0.11	7	224.59	1	2.42	37	52.21	1	endo	tr
<i>Diodia teres</i> (Walter) Small	0.73	23	63.93	2	0.41	8	109.77	1	0.28	38.33	106.66	3	auto	s
<i>Diospyros hispida</i> A. DC.	0.11	25.33	127.53	3					0.63	47.2	88.18	10	endo	ds
<i>Diplusodon virgatus</i> Pohl													anemo	tr
<i>Duguetia furfuracea</i> (A. St.-Hil.) Saff.	0.25	48.5	73.1	2	0.07	10.92	205.51	17	0.07	16.5	94.95	1	endo	ds
<i>Eremanthus erythropappus</i> (DC.) MacLeish	0.2	7	149.01	1	0.25	11.3	78.04	15	0.07	4	150.24	1	auto	ds
<i>Eriosema crinitum</i> (Kunth) G. Don	0.25	20.36	73.34	11	0.15	6	65.67	2	0.25	12.67	138.46	9	endo	s
<i>Erythroxylum campestre</i> A. St.-Hil.	0.26	7.33	125.19	9	0.07	10.92	205.51	17	0.15	11.04	346.36	14	endo	ds
<i>Eugenia angustissima</i> O.Berg					0.42	27	47.15	1					endo	ds
<i>Eugenia calycina</i> Cambess.					0.15	6	65.67	2					endo	ds
<i>Eugenia</i> sp.1 P. Micheli ex L.	0.22	5.83	48.03	3	0.02	8	120.12	1	0.21	72	190.03	2	auto	t
<i>Eugenia</i> sp.2 P. Micheli ex L.					0.12	10	149	1	0.12	10	149	1	auto	cl
<i>Eupatorium</i> sp. L.					0.1	26	53.55	1	0.1	26	53.55	1	auto	cl
<i>Galactia decumbens</i> (Benth.) Chodat & Hassl.	0.21	8	86.69	1	0.08	10	66.47	1					anemo	sb
<i>Galactia martii</i> DC.					0.13	10	262.28	1	0.3	36	151	2	epi	t
<i>Gomphrena macrocephala</i> A. St.-Hil.	0.95	6	104.94	1					0.41	21.5	60.48	2	epi	el
<i>Gymnopogon foliosus</i> (Willd.) Nees	0.4	22	227.14	1									anemo	ds
<i>Hyptis</i> sp. Jacq.													anemo	ds
<i>Jacaranda decurrens</i> Cham.													anemo	ds
<i>Jacaranda rufa</i> Silva Manso	3.07	16.33	94.02	3									anemo	ds



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