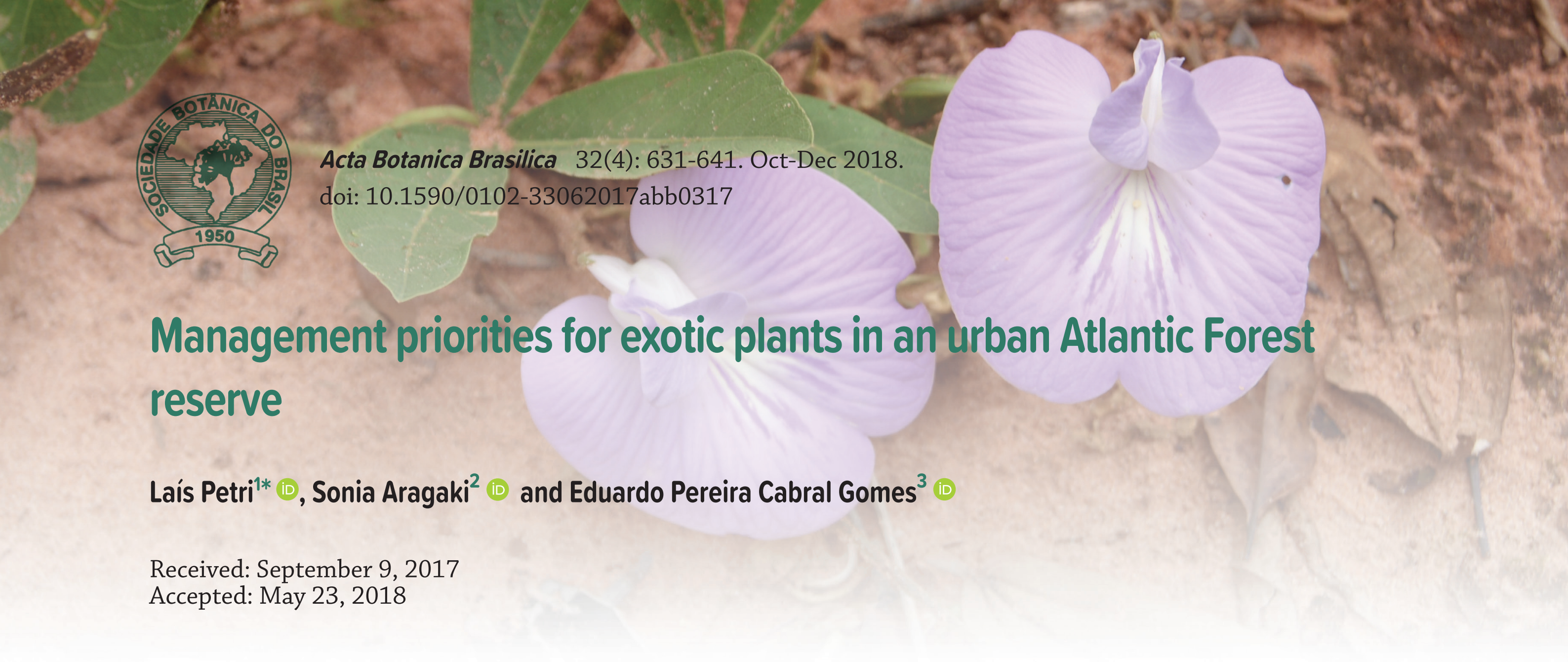

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Management priorities for exotic plants in an urban Atlantic Forest reserve

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ABSTRACT

Biological diversity is directly affected by alien species, even though the diagnosed impacts vary with scale. Early identification of the invasion of natural patches is essential for effective conservation actions. We aimed to determine the exotic plant species present inside Fontes do Ipiranga State Park (PEFI), an urban protected area located in the city of São Paulo, Brazil, and their relative proportions of occurrence in the native forest. Our goal was to use these data to assess their invasion status according to specific literature and to define management priorities for them. Therefore, we surveyed the presence of exotic plants within the canopy layer and understory of three native forest areas with different levels of disturbance. We found ten exotic plant species. The species found in both strata (60 %) were considered non-dominant ruderal. We assessed the density:coverage ratio to try to distinguish groups of priority, and found *Livistona chinensis*, *Archontophoenix cunninghamiana*, and *Syzygium jambos* to be classified as high priority for management. The early stage of the invasion process at PEFI indicates it is an area of high conservation value, and so we provide recommendations for management priorities prior to severe changes in the composition of the natural plant community.

Keywords: botanical garden, disturbance, invasion biology, invasion status, protected area, species population ecology

Introduction

Exotic species differ from native species in that they colonize regions where they originally would not have existed without intentional or accidental human transport (Richardson *et al.* 2000). The degree of establishment and success of an alien species, and thus their naturalization, in a new environment encompasses a gradient of the invasion process (Richardson *et al.* 2000). Invasive alien species are capable of constant propagation, viable population maintenance and efficient dispersion, which all favor the expansion of the area where they occur beyond the source of introduction, thus allowing them to enter new environments (Richardson *et al.* 2000). Due to these characteristics, invasive alien species are considered a serious environmental issue (Moro *et al.* 2012).

Indicators created to measure the rate of biodiversity loss have converged upon growing trends of pressure asserted by alien species on biological diversity (Butchart *et al.* 2010). The impact of exotic plant species on natural plant communities can increase biomass and primary production, although other variables, such as growth, abundance, diversity, and fitness of native plants, are harmed in the invasion process (Vilà *et al.* 2011). It is estimated that between 0.5 and 0.7 % of trees and shrubs are currently described by the literature as invasive (Pyšek *et al.* 2004). Since exotic species are not equally harmful, local scale assessments are needed despite

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their economic and environmental impacts (Rejmánek 2014). The most traditional definitions of the different paths in the invasion process differ by incorporating (Davis *et al*. 2000; Lockwood *et al*. 2007; Blackburn *et al*. 2014) or not (Richardson *et al*. 2000; Pyšek *et al*. 2004) the impacts alien species have on the native community for categorizing them as invaders.

Worldwide, cataloged species richness is 1.2 million species, however, considering just terrestrial species it is estimated that 86 % have not yet been discovered and described (Mora *et al.* 2011). South America has the greatest biological diversity in the world, housing around 20 % of all its flora and fauna (GISP 2005). Brazil is considered to have the richest flora of any country in the world, even though only approximately one-third of its species expected richness is known (Forzza *et al.* 2012). Biological invasion by exotic species has been one of the main factors for the loss of biodiversity globally (Kowarik 2011).

Given the characteristics of globally invasive alien species, comparative studies of floras can elucidate general patterns of plant invasions, thereby promoting a link between taxonomists and ecologists (Pyšek *et al.* 2004). Among researchers, whether to include exotic species data, or even highlight them as such, has been debatable, as Moro *et al.* (2012) explain when analyzing the scenario nationally. According to these authors, useful information is lost when naturalized and invasive species are excluded from published plant lists (Moro *et al.* 2012). Reviewing the Brazilian flora (BFG 2015), Zenni (2015) diagnosed the presence of non-native species in all biomes, but especially the Atlantic Forest where the largest number of naturalized species was found. Human actions and activities were directly correlated with the observed patterns of naturalization (Zenni 2015).

In Brazil, protected areas, whether federal, state or municipal, are also affected by exotic species (Ziller & Dechoum 2014). Early diagnosis of the establishment of exotic species in protected areas is a major factor and a starting point for accomplishing conservation purposes (Ziller 2009). An initial overview of biological invasion on a nation-wide scale has demonstrated the need for monitoring and management (Sampaio & Schmidt 2014), especially to avoid the introduction of invasive species into protected areas (Ziller 2009). For example, Fontes do Ipiranga State Park (PEFI) is a protected area within the Atlantic Forest biome that is surrounded by an urban matrix. PEFI was officially created in 1969, and the first study on vegetation structure within its boundaries was that of Vuono (1985), who indicated the presence of exotic tree species in the native forest. Eleven other studies involving floristics, phytosociology and population dynamics (Costa & Mantovani 1992; Nastri *et al.* 1992; Knobel 1995; Gomes & Mantovani 2001; Pivello & Peccinini 2002; Gomes *et al.* 2003; Davison 2006; 2009; Hirata *et al.* 2010; Tanus *et al.* 2012; Kondrat 2014) have since been carried out in different areas of PEFI (more than 10,000 individual plants sampled), all of which discriminated exotic species.

Considering this conceptual framework, and the current situation of the natural vegetation of PEFI, we aimed to determine the exotic plant species presently existing among the natural vegetation near the Botanical Garden of PEFI. The literature cites some exotic plant species as more frequently invasive than others. Thus, we also aimed to determine which of the exotic species found occurring in PEFI are found elsewhere around the world. Exotic plant species that occur only in the canopy layer are likely to have already been established, having become so only when the canopy was sufficiently open. Such a situation is compatible with a more disturbed environment and, considering the progression of restoration, exotic plant species are likely not to be able to change the status of the invasiveness. On the other hand, the presence of exotic plants in both strata (canopy layer and understory) of the forest, or only in the understory, may indicate a more continuous/recent disturbance. Therefore, we expect increasing similarity in the exotic plant species that occur in the canopy layer and the understory with increasing disturbance level. So, are the same exotic species present in the canopy layer and the understory? The investigation of exotic species in the understory strata can be indicative of either the regeneration of canopy layer individuals or the dispersal of species from other sites. In order to assess the characteristics of the populations of these species, we aimed to determine if the proportion of exotic species is the same between the canopy layer and the understory. Furthermore, we aimed to determine which exotic species should be key priorities for the allocation of resources for management in order to prevent the spread of invasion based on population ecology (Fig. 1).

Materials and methods

Study site

The study was conducted in the natural vegetation of Fontes do Ipiranga State Park (PEFI), a protected area located between 23°38’08’’S-23°40’18’’S 46°36’48’’W-46°38’00’’W in the city of São Paulo, Brazil, with elevations ranging between 759 and 837 m (Barbosa *et al.* 2002; São Paulo 2006). Currently, the total area of PEFI is around 490 ha, with the uncertainty being due to conflicting sources of information about the limits of the park. According to Köppen’s climatic classification, the region of the park is characterized as temperate climate Cwb, with an annual average temperature of 19.1 °C and an average annual rainfall of 1,540 mm (Santos & Funari 2002). Over eight decades of climate monitoring conducted by the Institute of Astronomy, Geophysics and Atmospheric Sciences of the University of São Paulo, which has a meteorological station inside PEFI, has recorded considerable climate

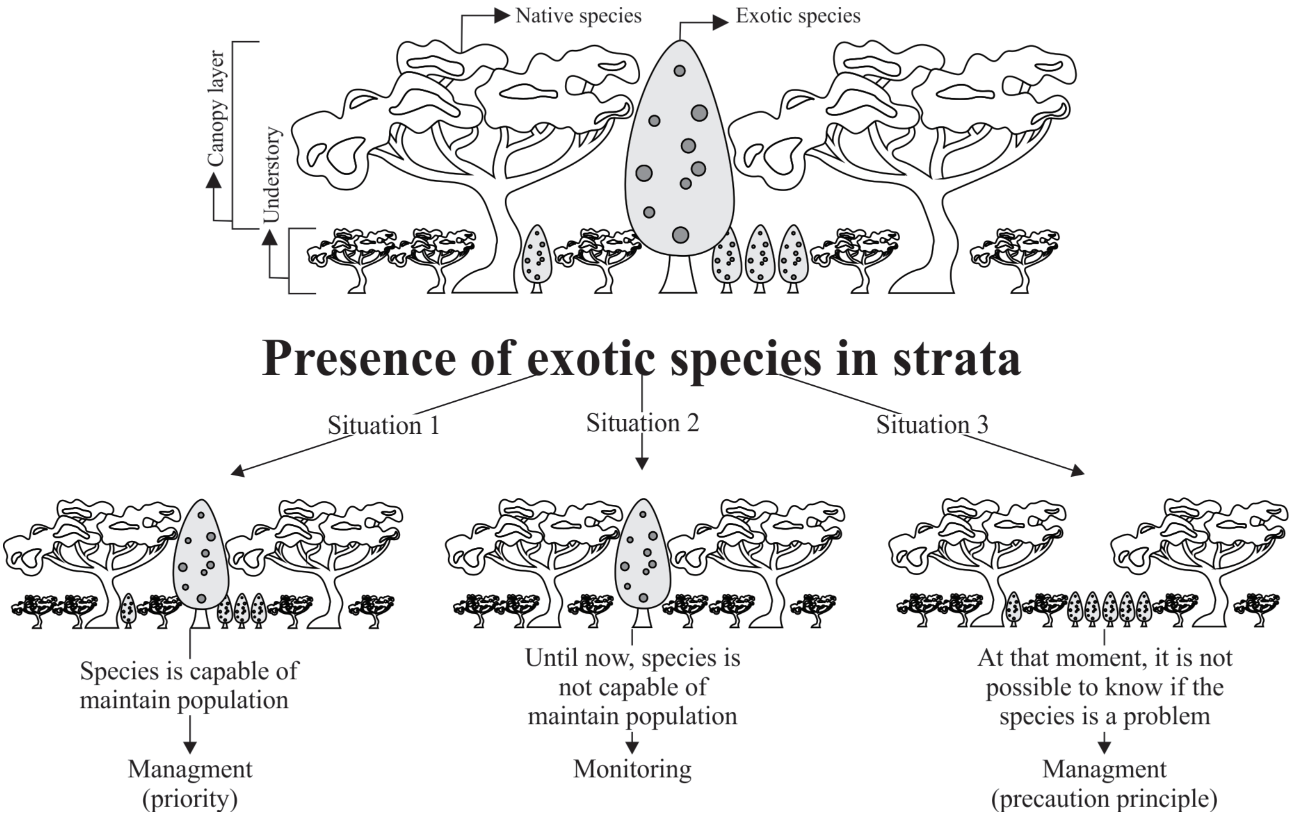


Figure 1. Conceptual model of a proposed framework for defining exotic plant management priorities in order to identify the potential for invasions to spread. In Situation 1, the exotic species is in both the canopy layer and the understory, suggesting that it has the capacity for maintaining a viable population, at least with surviving juveniles, and that it should be a priority species for management. In Situation 2, the exotic species is present only in the canopy layer, and without juveniles in understory. It is impossible to predict if such a species is incapable of leaving descendants or if they fail to survive, so monitoring is recommended. Situation 3 portrays a situation where only juveniles are detected. It is impossible to predict if this species will or will not turn into an environmental problem for the community, but its low density facilitates management at low cost. We also suggest that management of this exotic species follow the precaution principle.

change. The average annual and minimum temperatures for 2015 were, respectively, 20.4 ˚C and 16.3 ˚C, both the highest recorded since 1933, while the average annual maximum temperature of 26.8 ˚C was the second highest historical record (Camargo 2015). Rainfall exhibited a 30 % increase compared to the historical average (Camargo 2015). The predominant soil type in PEFI is Red Oxisol with considerable acidity — pH between 3.5 and 4.5 (Fernandes *et al.* 2002); the management plan indicates the presence of argisols and cambisols as well (São Paulo 2006). PEFI partially occupies the scarps of the Serra do Mar and the Serra da Mantiqueira, within the Planalto Atlântico (Atlantic Plateau; Almeida 1958 apud São Paulo 2006), comprising the Alto Rio Tietê basin and incorporating the headwaters of Riacho do Ipiranga (São Paulo 2006). The presence of headwaters in the area was the main justification for the creation of PEFI (São Paulo 2006).

PEFI is located in an Atlantic Forest patch, which suffered a reduction of 27 % between 1953 and 1994 due to urbanization both in the surroundings and within the park itself (Pivello & Peccinini 2002). It constitutes a vegetation island within an urban matrix, and so is affected by a variety of impacts. Most of the plant species present in PEFI are representatives of rainforest vegetation, with some elements of tropical semideciduous forest (Nastri *et al.* 1992; Barros *et al.* 2002). PEFI features over 1,500 native species of phanerogams (Barros *et al.* 2002), bryophytes (Visnadi 2015), and ferns and lycophytes (Hirai *et al.* 2016).

Field procedures

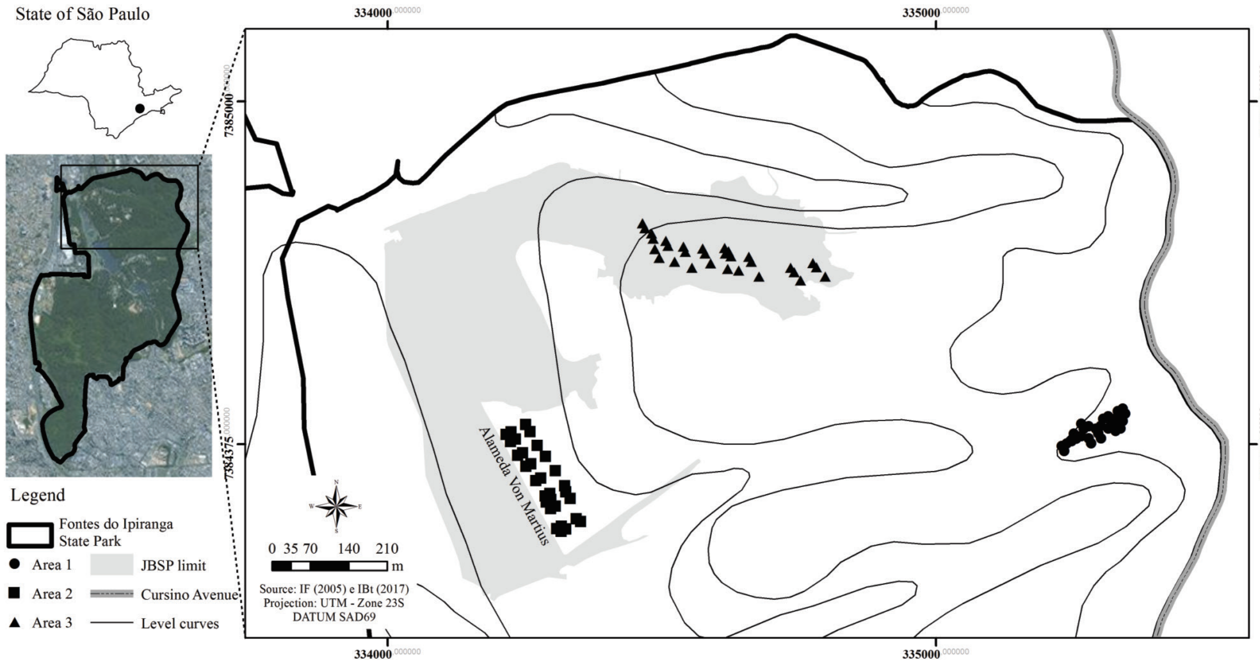
Considering that more than ten studies have sampled the vegetation of PEFI over the last 30 years, and that all of them indicated exotic species, their previous records were compiled to form a guide list (Tab. 1) for our sampling. Any additional exotic species found during our field procedures and taxonomic identifications were added to the list. We sampled the natural vegetation from September 2015 to October 2016 in 90 10 m x 10 m (100 m^2) plots for a total sampled area of 9,000 m^2. The plots were equally distributed among three different native forest areas (i.e., 3,000 m^2 in each area) (Fig. 2). Historical land-use mapping from 1953 to 1994 (Peccinini & Pivello 2002), revealed that the vegetation patch that contains Area 1 remained categorized as "heterogeneous canopy forest", with only the height of vegetation changing over time. Area 1 has not suffered any direct disturbance such as fire, logging, or understory clear-cutting from 1994 to the present. Consequently, Area 1 has been without severe anthropic disturbances for at least 64 years, although it is just 150 m from the park perimeter (east boundary with Cursino Avenue). This low level of perturbation has been corroborated by recent studies on nutrient cycling (Moraes 2002; Santos 2014; Bazi & Gomes 2015; 2016; Vieira 2015) and plant community dynamics (Davison 2006; 2009; Carvalho *et al.* 2011; Kondrat 2014).

The road Alameda Von Martius, which belongs to the Botanical Garden (JBSP), an administrative unit inside PEFI, borders Area 2. The area is considered to have intermediate disturbance, with its floral composition and plant structure being consistent with that of disturbed environments (when studied from the edge to the forest inner by Tanus *et al.* 2012). Finally, Area 3 has a higher level of disturbance than the other two areas. It is located in the forest of JBSP, and is adjacent to Terra Batida Trail, which is used daily by visitors and for environmental education. The floristic patterns of Area 3 were partially investigated by Hirata *et al.* (2010), who indicated that the use this trail for public purposes influences plant species establishment. Therefore, these three areas represent the entire disturbance gradient of the natural area of PEFI.

Sampling of exotic individuals comprised two size classes reflecting occurrence in the understory or the canopy layer. In the understory, all individuals with height > 1 m, and diameter at breast height (dbh) < 4.8 cm were counted but only the exotic species were identified, making it possible to calculate the percentage of exotics in the regenerating layer of each plot. For the canopy layer (dbh > 4.8 cm), all individuals (except climbers, and epiphytes)

Table 1. Exotic plant species with specific previous records of occurrence in Fontes do Ipiranga State Park, São Paulo, SP. The compilation served as a guide list for field searches during sampling.

Family	Exotic species	Reference
Arecaceae	*Archontophoenix cunninghamiana* H.Wendl. & Drude	Catharino & Silva (2007), Hirata *et al.* (2010), Tanus *et al.* (2012)
Rubiaceae	*Coffea arabica* L.	Catharino & Silva (2007), Kondrat (2014)
Asparagaceae	*Dracaena* sp.	Davison (2009)
Rosaceae	*Eriobotrya japonica* (Thunb.) Lindl.	Catharino & Silva (2007), Tanus *et al.* (2012)
Rhamnaceae	*Hovenia dulcis* Thunb.	Catharino & Silva (2007)
Arecaceae	*Livistona chinensis* (Jacq.) R.Br. ex Mart.	Catharino & Silva (2007)
Pittosporaceae	*Pittosporum undulatum* Vent.	Vuono (1985), Nastri *et al.* (1992), Catharino & Silva (2007)
Myrtaceae	*Psidium guajava* L.	Peccinini & Pivello (2002)



Source: IF (2005) e IBt (2017)
Projection: UTM - Zone 23S
DATUM SAD69

Figure 2. Map of the studied protected area. Location of distributed plots in the three sampled areas (Area 1 = lowest disturbance level, Area 2 = intermediate disturbance level, Area 3 = highest disturbance level) of natural vegetation inside Fontes do Ipiranga State Park, São Paulo-SP. JBSP = Botanical Garden of São Paulo.

were sampled. For all species (native or exotic) in canopy layer dbh was measured and plant material collected for species identification. All the botanical material collected was identified to the species level, whenever possible, with support from Núcleo de Pesquisas e Curadoria do Herbário SP of the Instituto de Botanica. We incorporated one-dried voucher of each species into the collection of SP Herbarium. The coverage of each exotic species was estimated by calculating the canopy area of each individual by approximating it into an ellipse shape, i.e. the product of the semi-major axis and the semi-minor axis of canopy, multiplied by pi.

Data analysis

Differences in species richness among sampling units (Areas 1, 2 and 3) were compared using individual rarefaction (1,000 permutations), with the respective 95 % confidence interval, to control for differences in sampling individual's effort (Krebs 1989). We evaluated diversity among the three areas using the diversity profile, which allows simultaneous analysis of species richness and relative abundance of each species (Tothmeresz 1995).

For the canopy layer, the usual quantitative descriptors (absolute and relative abundances, relative frequency and dominance, and importance and cover values) of vegetation structure were performed, including the entire community, using Fitopac 2.1.2.85 (Shepherd 2010). The definition considered for a rare species was that of Martins (1991), who defined a rare species as having a unique individual in the sample. For the understory, we calculated the proportion of exotic individuals relative to native plants. In order to test for significant differences in the densities of exotic species among the three areas, the non-parametric Kruskal-Wallis test was used, and when there was a significant difference ($p < 0.05$) the Mann-Whitney pairwise (non-parametric) *post-hoc* test was applied. With the purpose of identifying significant differences in the density of each exotic species between the canopy layer and the understory in all areas surveyed, the Mann-Whitney pairwise (non-parametric) test was applied. The usual tests for normality (Shapiro-Wilk) and homogeneity of variance (Levene) were performed *a priori*, but even with transforming the data it was not possible to use parametric tests. We performed all analyses in R (R Core Team 2016), mainly using the packages "iNEXT", "entropart", "car" and "ggplot2". The script is available in Text S1 in supplementary material.

The definition of invasion status for the exotic species followed the key proposed by Durigan *et al.* (2013). However, in order to avoid conflicts in the classification of management priorities between the proposed population ecology conceptual framework (Fig. 1), and the invasion status from the criteria defined by Durigan *et al.* (2013), we performed a regression analysis to investigate the effect of exotic individuals' density on exotic coverage. We expected that species with both higher density and coverage would be indicated as a priority for management, especially in cases when an exotic species was present both in the canopy layer and in the understory but was classified as non-dominant ruderal, which is typical of early invasion. The geographical regions of origin of the alien species were determined using the national database of invasive alien species I3N Brazil (Instituto Hórus 2017).

Results

With the number of individuals fixed, the rarefaction curves for the presence of exotic species in the community (Fig. 3A) showed no differences in species richness among the

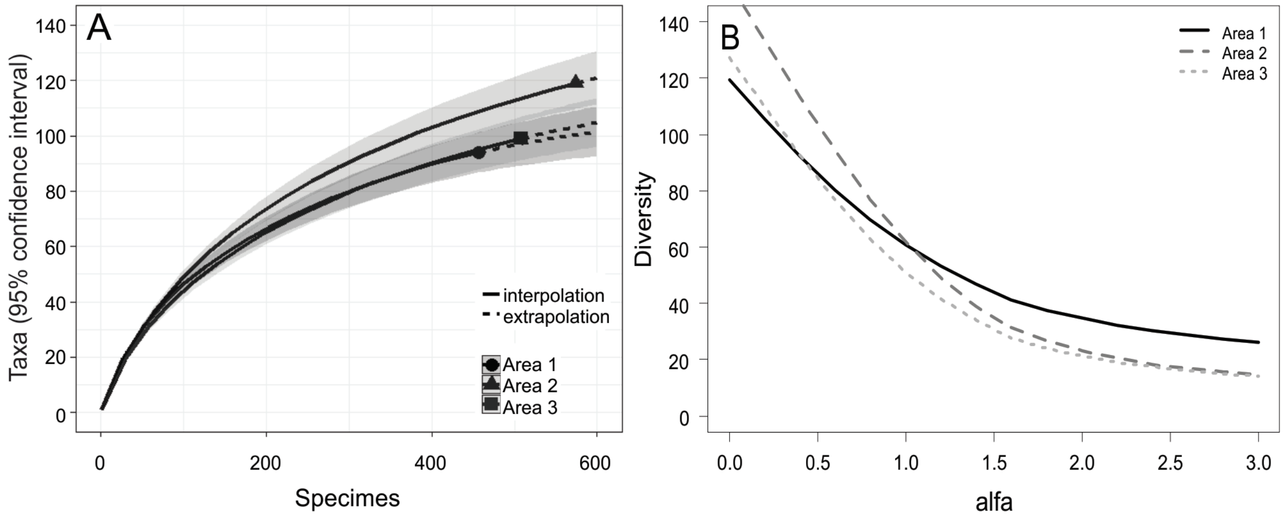


Figure 3. Analysis comparing the three sampled areas (Area 1 = lowest disturbance level, Area 2 = intermediate disturbance level, Area 3 = highest disturbance level) considering the entire community (i.e., native and exotic species). **A.** Individual rarefaction curves with 1,000 simulations. **B.** Diversity profiles comparing the sampled areas.

areas due to overlapping confidence intervals. However, Area 2 (intermediate level of disturbance) had higher richness with a sampling effort of 400 specimens, indicating a point at which sampling effort could reveal significant differences in observed richness from the other areas. In the diversity profile (Fig. 3B), the omission of the evenness component showed that Area 2 was always richer. The presence of alien species afforded a higher richness to Area 3 (higher level of disturbance) when compared to Area 1 (lowest level of disturbance) (alpha = 0). When evenness is included, that is by including the relative abundance of species by using indexes equivalent to Shannon's (alpha = 1) and Simpson's (alpha = 2) indexes, the diversity profile curves intersect and Area 1 has the greatest diversity.

In terms of the complete community (native and exotic species) in the canopy layer, Area 1 had 476 individuals of 103 species from 39 families (Tab. S2 in supplementary material); Area 2 had 587 individuals of 130 species from 41 families (Tab. S3 in supplementary material); and Area 3 had 521 individuals of 105 species from 40 families (Tab. S4 in supplementary material). There were no significant differences in basal area (One-way ANOVA, $F = 0.08213$, $p > 0.05$), and density of canopy individuals (One-way ANOVA, $F = 2.48$, $p > 0.05$) among sites. Considering all areas together as a unit, Myrtaceae was the family with the greatest number of individuals (194) and species (35) and, together with Lauraceae, Rubiaceae, Fabaceae, Melastomataceae, and Euphorbiaceae, comprised 50.4 % of all species identified. A total of 32.2 % of all the species were considered rare. None of the exotic species were among the plants with the highest importance values (IV).

A total of 10 exotic species were sampled, and the density of exotic individuals per hectare in the understory was four times higher than that of the canopy layer (Tab. 2). The loquat (*Eriobotrya japonica*) and *Pinus* sp. were exclusively found in the canopy layer. In contrast, *Dracaena fragrans* and *Livistona chinensis* were found only in the understory. Three of the exotic species found were not on the guide list (Tab.1), and thus recorded in PEFI for the first time: *Caryota urens*, *Persea americana* and *Syzygium jambos*. The rose apple (*S. jambos*) was the species with the greatest density of individuals per hectare, with 94.6 % of them being in the understory. Area 1 had a lower density of exotic individuals per hectare (43.4 indiv. ha^{-1}) (Kruskal-Wallis $H = 9.447$ $p < 0.05$) than Area 2 and Area 3 (Tab. 2); only the exotic species *Coffea arabica* and *L. chinensis* occurred in Area 1, where they were exclusive to the understory. These species were also found in Area 2 and Area 3. Area 3 had the highest richness of exotic species (seven species). The rose apple (*S. jambos*) was present exclusively in disturbed areas in both Area 2 and Area 3, but with greater abundance in the former. Comparing the densities of each exotic species distributed in both strata, there was a significant difference in densities between the canopy layer and the understory for *C. arabica* (Mann-Whitney $U = 3528.5$ $p < 0.05$), *L. chinensis* (Mann-Whitney $U = 3580.5$ $p < 0.05$), and *S. jambos* (Mann-Whitney $U = 3516$ $p < 0.05$).

Considering the invasion status of the exotic species, *E. japonica* and *Pinus* sp. were categorized as casual alien species since they were not leaving persistent descendants. *Archontophoenix cunninghamiana*, *C. urens*, *D. fragans*, *P. americana*, *Pittosporum undulatum*, and *S. jambos* were classified as non-dominant ruderal species since they were only present in one or both of the areas (Area 2 and Area 3) considered to have some level of disturbance and do not dominate the community there. The species *C. arabica* was classified as a non-dominant invader since it is present in Area 1, but has no detectable direct effect on the native community. The species *L. chinensis* was classified as a dominant invader, and also occurred in the least disturbed area (Area 1), as did *C. arabica*. In addition, less regeneration of native species was diagnosed under de

Table 2. Density (individuals.hectare-1) of exotic species in the canopy (dbh > 4.8 cm) layer and understory (height > 1 m and dbh < 4.8 cm) for each area (Area 1 = lowest disturbance level, Area 2 = intermediate disturbance level, Area 3 = highest disturbance level) in Fontes do Ipiranga State Park, São Paulo, SP. Invasion status (Durigan *et al.*, 2013): casual alien species (CaAl), non-dominant ruderal (NDoRu), non-dominant invader (DoIn), dominant invader (DoIn). Origin of exotic species: I3N Brazil database of invasive alien species. Mann-Whitney pairwise (non-parametric) test.

Family	Exotic species	Origin	Density (indiv.ha^{-1})					Invasion status
			All areas		All size classes			
			Canopy	Understory	Area 1	Area 2	Area 3	
Arecaceae	*Archontophoenix cunninghamiana* H.Wendl. & Drude	Australia	6.7^{a}	5.6^{a}	-	-	36.7	NDoRu
Arecaceae	*Caryota urens* L.	Asia	2.2^{a}	1.1^{a}	-	-	10.	NDoRu
Rubiaceae	*Coffea arabica* L.	Africa	3.3^{a}	32.2^{b}	36.7	20.0	50.	NDoIn
Asparagaceae	*Dracaena fragrans* Ker Gawl.	Africa	0.0^{a}	4.4^{a}	-	-	13.3	NDoRu
Rosaceae	*Eriobotrya japonica* (Thunb.) Lindl.	Asia	1.1^{a}	0.0^{a}	-	3.3	-	CaAl
Arecaceae	*Livistona chinensis* R.Br.	Asia	0.0^{a}	24.4^{b}	6.7	10.0	56.7	DoIn
Lauraceae	*Persea americana* Mill.	Central America	3.3^{a}	5.6^{a}	-	26.7	-	NDoRu
Pinaceae	*Pinus* sp.	North America	1.1^{a}	0.0^{a}	-	3.3	-	CaAl
Pittosporaceae	*Pittosporum undulatum* Vent.	Australia	4.4^{a}	2.2^{a}	-	-	20.0	NDoRu
Myrtaceae	*Syzygium jambos* (L.) Alston	Asia	2.2^{a}	38.9^{b}	-	96.7	26.7	NDoRu
Total					43.4^{a}	160.0^{b}	213.4^{b}	

L. chinensis individuals, possibly due to shading in Area 3. Applying the conceptual framework proposed in Fig. 1, the management indications for these species would be: *E. japonica*, and *Pinus* sp. as low priority and *L. chinensis* as high priority. The other species considered as low (the remaining) or intermediate (*C. arabica*) priority by their invasion status, we would consider as high priority for management following the conceptual framework proposed in Fig. 1. Even though the adjusted linear regression had a low coefficient ($R^2 = 0.18$), two distinct groups could be highlighted: *A. cunninghamiana,* and *L. chinensis* (the latter as expected) with higher coverage values, and *C. arabica*, and *S. jambos* with higher values for density:coverage ratio (Fig. 4).

The geographic origins of the studied exotic species included Asia, with four species, followed by the Australia and Africa, with two species each, and then Central and North America, with one species each.

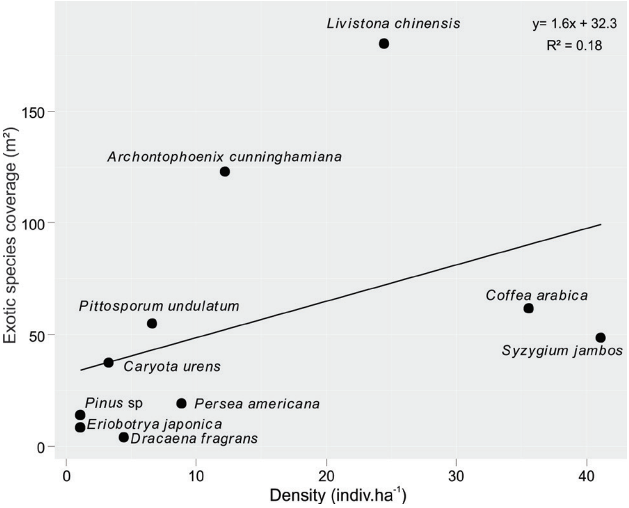


Figure 4. Visual representation of the ratio between coverage and density of exotic species. Estimated exotic species coverage as a function of individuals per unit area revealing important species in need of priority management among others diagnosed as non-dominant ruderal (*Archontophoenix cunninghamiana* and *Syzygium jambos*).

Discussion

The most frequent geographic regions for the origin of the exotic species encountered in the present study are compatible with the literature. Asia can be considered the main source region for exotic tree species in the world, followed by Australia. Africa and North and Central America are also characterized as important sources of non-native species (Rejmánek 2014). North America and Europe are the continents that provide more species of Pinaceae (Rejmánek 2014), which includes the genus *Pinus*. A global review of invasive trees and shrubs found that the genus *Pinus* is invasive in 11 of the different geographic regions defined by Richardson & Rejmánek (2011), among which South America is included. The Asian continent, the source region for *C. urens* and *L. chinensis*, is the greatest source of Arecaceae in the present study (Rejmánek 2014). Among the most widespread invasive alien plant species in the world, *P. undulatum* and *S. jambos* are in at least six of the geographic regions studied by Richardson & Rejmánek (2011), and both have invaded South America.

The sweet pittosporum, *P. undulatum*, is an Australian species, where it occurs along the coastline and in mainland mountains (Souza *et al.* 2016). The species is considered invasive worldwide. In Brazil, the species is invasive in forests of the state of Paraná (Blum *et al.* 2005; Mielke *et al.* 2015), while in São Paulo it is considered as subspontaneous (Garcia & Pirani 2001), as exotic (Almeida *et al.* 2010), as with invasive potential (Nascimento *et al.* 2011a), and as a dominant invader (Souza *et al.* 2016), with documented dispersion by frugivorous birds (Campagnoli *et al.* 2016). This species was categorized as a non-dominant ruderal species in our study, however, according to the literature, it is considered invasive in several places. In PEFI, large numbers of individuals of the species are known to occur in other non-sampled vegetation areas (e.g., in the vegetation north of the JBSP visitation area), making population monitoring extremely important.

The rose apple, *S. jambos*, is considered a dominant invader of 27 of 30 groups of oceanic islands studied by Kueffer *et al.* (2010), appearing on massifs. These include, for example, the Galapagos Archipelago (Watson *et al.* 2010) and the Pitcarin Islands, British Territory in Polynesia, where it was introduced two centuries ago and has already replaced altitudinal and lowland forests (Kingston & Waldren 2003). In Brazil, *S. jambos* was found to have a high importance value both in the understory (Santiago *et al.* 2014) and in the canopy layer of an urban forest in the state of Minas Gerais (Fonseca & Carvalho 2012). In São Paulo the species is considered as an exotic (Almeida *et al.* 2010), and as a non-dominant invader in Alberto Löfgren State Park (Souza *et al.* 2016). Its characteristics confirm the recent invasive diagnosis, as this species was the most abundant exotic found in PEFI, and so we consider it a priority for management.

Also in Brazil, the invasion and dominance by the Australian palm, *Archontophoenix cunninghamiana* are well documented in the forest of Cidade Universitária "Armando Salles de Oliveira" – CUASO – (Dislich *et al.* 2002; Christianini 2006; Mengardo *et al.* 2012; Mengardo & Pivello 2014), and has been recorded by other population studies (Nascimento *et al.* 2011b; Garcia & Pirani 2001; Souza *et al.* 2016). Another palm tree identified in the present study, *Caryota urens*, was classified as invasive in Manipur, India (Singh *et al.* 2015). Finally, the Chinese fan palm (*L. chinensis*) is considered to be one of the most documented invasive palms (Meyer *et al.* 2008). Of the three invasive palm trees found in PEFI, *L. chinensis* was the most abundant in the understory and exhibited invasive attributes, while *A. cunninghamiana* had a high density:coverage ratio.

The coffee, *Coffea arabica*, can be considered invasive in the Pacific Islands (Meyer 2000) and Hawaii (Mascaro *et al*. 2008), and an exotic ornamental without indications of naturalization in South Africa (Foxcroft *et al*. 2008). In Brazil, this species was found to occur in low density in the regenerating layer of a tropical semideciduous forest in Juiz de Fora, Minas Gerais (Santiago *et al*. 2014), and in São Paulo, where it has been characterized as having invasive potential (Nascimento *et al*. 2011a) and as the dominant invader (Souza *et al*. 2016). The species has also been recorded in the state of Rio de Janeiro, the location for the introduction of coffee to the country in the middle of the 18th century (Silva 2006). Coffee cultivation spread by planting in farms and backyards in São Paulo (Silva 2006), and often remains present in the most remote regions of small urban centers. This information was confirmed by Article 3rd, Law No. 2020 (Brasil 1924 apud Silva 2006), enacted with the aim of rapidly diagnosing it as a major pest coffee, and emphasizing the presence of isolated or monoculture plantations in farms, ranches, sites, gardens or orchards (Silva 2006). Thus, we believe that *C. arabica* was present in PEFI before the park was even created, given the history of its foundation from the expropriation of land. Thus, this species is not considered an immediate issue for the natural vegetation of PEFI.

The dispersion and presence of avocado, *P. americana*, can be associated with anthropogenic activities (Souza *et al*. 2016) and structures (Alston & Richardson 2006). This species was considered invasive in Hawaii (Mascaro *et al*. 2008) and possibly naturalized in South Africa (Alston & Richardson 2006; Foxcroft *et al*. 2008). In São Paulo, *P. americana* is known to occur in a number of different parks (Garcia & Pirani 2001; Almeida *et al*. 2010; Souza *et al*. 2016). Similarly, the dispersion of *E. japonica* is human mediated (Souza *et al*. 2016), and in South Africa it is considered the invasive species most frequently used for medical purposes (Maema *et al*. 2016). In Brazil, this species has been recorded as invasive in Paraná (Blum *et al*. 2005), while in São Paulo it has already been sampled among natural vegetation (Garcia & Pirani 2001; Catharino *et al*. 2006; Almeida *et al*. 2010; Souza *et al*. 2016). Considering that PEFI is a protected area and among the categories of greater restriction, the administrative management of the park through preventive actions connected to public use indirectly controls human-associated dispersion of these species. This situation corroborates the low management priority for both *P. americana* and *E. japonica*.

The corn plant, *Dracaena fragrans*, is considered invasive in Costa Rica (Morales 2015) and occurs along trails in Mount Halimun-Salak National Park in Indonesia (Kudo *et al*. 2014). In Brazil, the only invasion record found for the species is in Tijuca National Park, Rio de Janeiro, where it was diagnosed as a competitive and aggressive invader (Ribeiro 2006). Considering the punctual nature of the invasion records for this species throughout the world and in Brazil, non-dominant ruderal invasion status with low priority of management seems consistent with PEFI's reality.

A single individual of the genus *Pinus* was found in the present work, however, this genus is commonly found in protected areas in South and Southeast Brazil. In those places, its invasion is generally enhanced by the presence of adjacent reforestation plots or typically open native vegetation areas such as Itapeva Ecological Station (Almeida *et al*. 2010), Angatuba Ecological Station (Monteiro *et al*. 2009), and Itirapina Ecological Station (Zanchetta & Diniz 2006). Likewise, the probable propagule source of this genus in PEFI is an arboretum adjacent to Area 2. Since no juvenile plants were found in the understory, even though the adult tree sampled inside the forest was reproductive, low priority in management with periodic monitoring should be a sufficiently careful strategy for PEFI. It should be noted that the literature search for evidence of occurrence and/or invasion throughout the world and in Brazil has not been exhaustively discussed here and that the records presented are simply exemplary.

The exotic species diagnosed in PEFI vegetation did not have high important values (IVs) in any of studied areas. Consequently, the exotic species are not a dominant component of the community structure. In contrast, in a secondary succession semideciduos forest, Moreira & Carvalho (2013) diagnosed *Pinus elliottii* Engelm. as being among the five species with highest IVs, where its presence is due to propagule pressure, and *S. jambos* as opportunistic in places with anthropic disturbance. Both of these species were identified in our study (for Pinaceae taxonomic identification stopped at the level of genus). In a study at Alberto Löfgren State Park, a protected area in São Paulo that began as a botanical garden, all the exotic species found, with the exception of *C. urens*, were also found in PEFI (Souza *et al*. 2016). About 29 % of the richness found at Alberto Löfgren State Park was exotic, and of these *P. undulatum*, the genus *Pinus* and the Australian palm *Archontophoenix* spp. were among the 12 plants with the highest IVs (Souza *et al*. 2016). When combined with previous studies of vegetation structure in PEFI, *A. cunninghamiana* is the most abundant, with a total of 45 individuals (Hirata *et al*. 2010; Tanus *et al*. 2012), followed by *P. undulatum* with eight (Vuono 1985; Nastri *et al*. 1992), *Dracaena* sp. with five (Davison 2009), *C. arabica* with three (Kondrat 2014), and *E. japonica* with one (Tanus *et al*. 2012) (the same individual counted by us).

Disturbance has a strong influence on the process of invasion by exotic species (Lake & Leishman 2004). If the typical vegetation regime of disturbance an area is maintained, it can serve to decrease the propensity for invasion (Alpert *et al*. 2000), but if it is atypical, it can become decisive in the successful invasion of non-native species and their impact on the native plant community

(Lake & Leishman 2004; Hansen & Clevenger 2005; Jauni *et al.* 2015). The differentiation and categorization of exotic species invaders as those that invade undisturbed natural ecosystems and those that survive in disturbed areas as ruderals by Durigan *et al.* (2013), is consistent with our purposed framework based on population ecology. This distinction seems to facilitate a process of initial filtering of all exotic species resident in a specific region. From this confluence of interpretations of the presence of exotic species in the native plant community, specifically in PEFI, *L. chinensis* is considered to be of high priority for management, while *E. japonica* and *Pinus* sp. as low priority.

As highlighted by Durigan *et al.* (2013), their dichotomous key is effective in cases where the establishment of exotic species has already occurred. We noticed that the low proportions of exotics in the community concentrate most species in a unique category, as does our population interpretation model, though its objective was to define and refine priorities. Consequently, we suggest that the density:coverage ratio for exotic species is both an easy and fast measurement to make during field data collection. Furthermore, it seems to visually separate groups and serves as an elucidative alternative for distinguishing the priorities of exotic species in such cases. Considering this new approach, *A. cunninghamiana*, *C. arabica*, and *S. jambos* were added as high priority species for management actions. As discussed above, the presence of coffee is probably due to the historical land use of the areas that were decreed as PEFI. This species has been present in PEFI for at least 48 years and, until now, in low densities compared to the native community. Hence, we suggest *C. arabica* to be monitored instead of managed at the present moment. We indicate periodic monitoring for all other exotic species not classified as high priority.

The invasion process detected at PEFI is in its early stages allows for more opportunities with regard to making decisions about management based on the actual scenario (Crooks 2005). However, the low abundance of exotic species should not be ignored. The initial stage of the invasion of PEFI forests must be addressed with responsibility and in a timely manner in order to maintain the integrity of this important Atlantic Forest patch immersed in an urban matrix. Thus, the presented invasion panorama makes it possible to identify some invasive exotic species as priorities for management, and assist decision-making and allocation of financial resources.

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