Landscape context of plantation forests in the conservation of tropical mammals

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A B S T R A C T

Plantation forests have been expanding in many tropical and subtropical environments. Howerver, even when they replace less wildlife friendly land uses such as pastures and annual crops, the biodiversity levels of pristine natural habitats often have not been recovered. Here we addressed how the landscape context of plantation forests located in South-eastern Brazil affects species richness and community resilience of medium and large size mammals. The area covered by native habitat fragments surrounding plantation forests is positively related to functional richness, including the presence of species more vulnerable to extinction in fragmented landscapes. In addition, the degree of aggregation of plantation forest stands is negatively related to more vulnerable species. No primates were recorded in our seven plantation forest sites (ranging from 272 to 24,921 ha), even when they were seen in native habitat fragments adjacent to commercial tree stands. Two invasive species (*Sus scrofa* and *Lepus capensis*) were recorded in four plantation forest sites. The impoverishment of fauna in plantation forests is due to two factors. First, plantation forests generally are structurally simplified habitats when compared to highly diverse tropical forests. Secondly, the isolation from habitat fragments which act as source of individuals in the landscape precludes the establishment of individual in plantation forest. We also highlighted the management practices to improve the complexity of vegetation in commercial tree stands should be taken cautiously, insofar as reduced productivity per area entails a greater demand for land. Thus, an alternative would be intensify the management of the commercial tree stands for wood production together with the restoration of adjacent areas set aside to conservation and native habitat fragments protection.

1. Introduction

The conversion of natural environments and land use intensification in human-dominated landscapes has led to striking changes in the Earth's surface (Foley et al., 2005), resulting in an unprecedented global species loss and an increasing number of threatened species (Barnosky et al., 2011; Hoffmann et al., 2010; Pimm et al., 2014). Within this context, further loss in biodiversity and ecosystems services could be reduced through the suitable management of landscapes which have undergone anthropogenic changes (Gardner et al., 2009). However, the development and adoption of new practices of landscape management is a complex task, so that in recent years it has taken the shape of debate about land-sharing and land-sparing management strategies (Fischer et al., 2014). Landscape management should consider the population fluctuations of each species (Vandermeer & Carvajal, 2001), the species-specific responses to the structure and configuration of the landscape (Lindenmayer, Cunningham, Donnelly, & Lesslie, 2002; Tscharntke et al., 2012) and their resilience to habitat changes (Ewers & Didham, 2006; Umetsu, Metzger, & Pardini, 2008). Hence, the landscape context, i.e. how both structure and configuration of habitat fragments modulate the interaction of the species with surrounding human land-uses, is an important focus of ecological research in human-dominated landscapes to understand the driving forces of persistence or local extinction (Daily, Ceballos, Pacheco, Suzán, & Sánchez-Azofeifa, 2003; Fischer, Lindenmayer, & Manning, 2006; Lindenmayer et al., 2008).

Among the several human land-uses, the functional role of plantation forests as it relates to biodiversity conservation has also been vigorously debated (Brockerhoff, Jactel, Parrotta, Quine, & Sayer, 2008). Plantation forests are expanding worldwide as result of an increasing demand for timber and pulp, and already represent about 7.3% of the world's forest cover (FAO, 2015). In Brazil, the area planted with
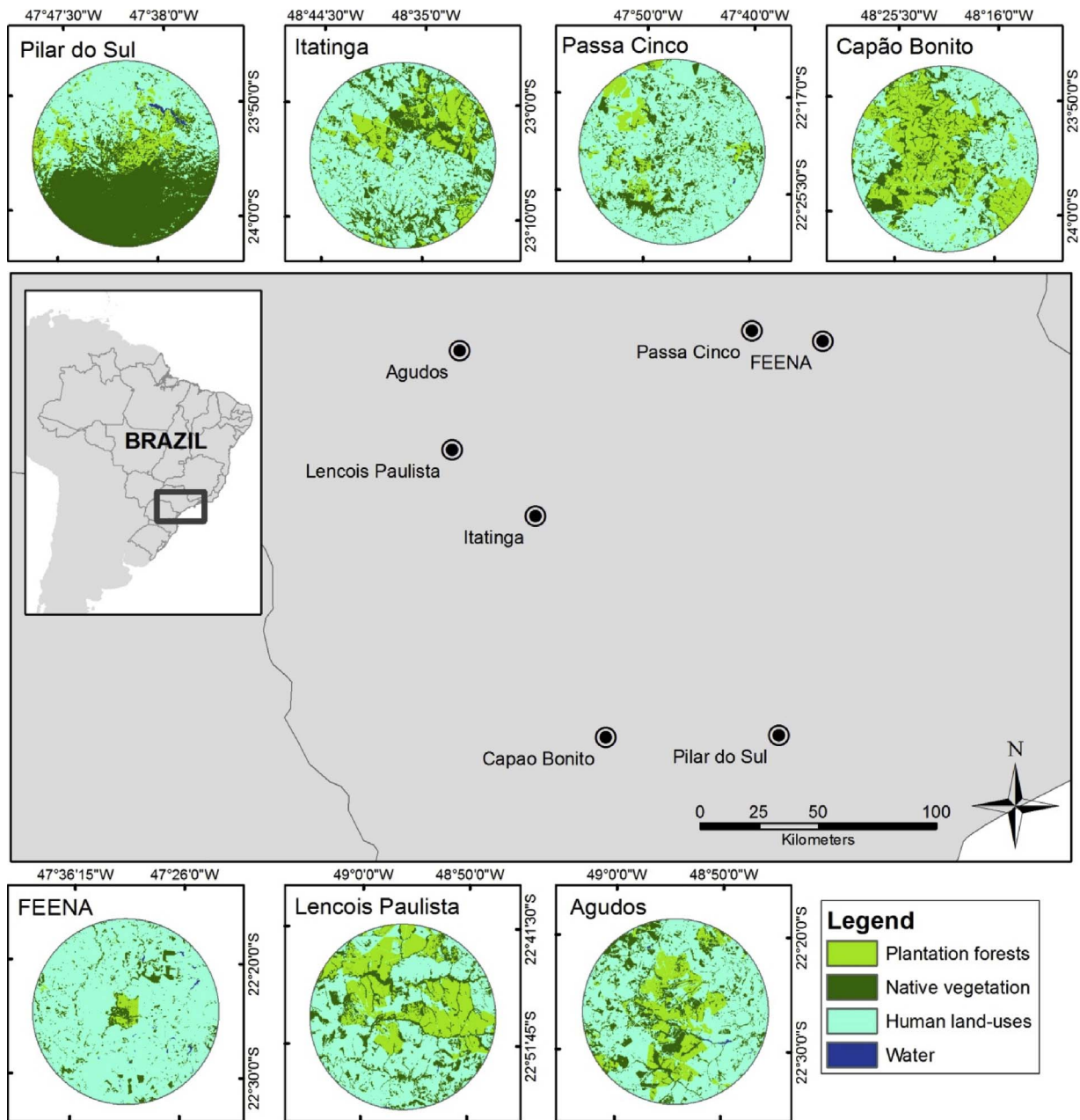
Fig. 1. Location of the seven plantation forests sites in São Paulo State, southeastern Brazil.

exotic *Eucalyptus* and *Pinus* trees reached more than 6.6 million hectares in 2012 (∼2.5% of country's territory), and only in the São Paulo state, forest plantations covered about 1,186,497 ha or 17.8% of state's territory (ABRAF, 2013).

Although management practices for conservation of biodiversity differ widely depending on target-species, the role of plantation forests could be effectively complementary to protected areas and large tracts of pristine habitats (Barlow, Gardner, et al., 2007; Brockerhoff, Jactel, Parrota, & Ferraz, 2013). Usually, plantation forests seem to hold a subset of the species assemblage found in undisturbed native habitats, even where the latter is the dominant land cover in the landscape (Barlow, Gardner, et al., 2007; Lindenmayer & Hobbs, 2004). On the other hand, plantation forests can improve connectivity between habitat patches and provide a sub-optimal habitat for the biota in highly fragmented landscapes (Brockerhoff et al., 2008; Hartley, 2002). Thus, plantation forests could play a substantial role in biodiversity conservation if their management reduces the detrimental effects of habitat

loss and fragmentation in both local and landscape scales (Brockerhoff et al., 2013).

The negative effects of habitat fragmentation are observed at multiple spatial scales. Changes in the structure and configuration of landscapes surrounding fragments affect the persistence of several taxa such as birds, mammals and others (Andrén, 1994; Mazerolle & Villard, 1999). In studies at the fragment scale, the best predictors of changes in patterns of species abundance are given by inter-patch variables, such as Fragment Size and Edge Area, whereas landscape-scale variables such as Isolation are better predictors of species occurrence (Thornton, Branch, & Sunquist, 2011). Thus, the occurrence of species in plantation forests could be an effect of landscape context, particularly, the distribution of native habitat patches and plantation stands in a given landscape.

The effectiveness of plantation forests for biodiversity conservation has been poorly studied when considering the influence of landscape attributes surrounding the commercial tree stands and vice versa. This

is particularly true for Atlantic forest domain in Brazil, where only about 12% of the original forest cover remains (Ribeiro, Metzger, Martensen, Ponzoni, & Hirota, 2009). The research effort has been focused almost exclusively in the comparison of species richness and assemblage composition found in native habitats and commercial trees stands. These studies have demonstrated the negative consequences of habitat simplification from commercial tree monocultures in relation to high diverse native tropical forest for birds (Marsden, Whiffin, & Galetti, 2001; Volpato, Prado, & dos Anjos, 2010; Zurita, Rey, Varela, Villagra, & Bellocq, 2006), mammals (Dotta & Verdade, 2011; Lyra-Jorge, Ciocheti, & Pivello, 2008; Martin, Gheler-Costa, Lopes, Rosalino, & Verdade, 2012; Stallings, 1991), spiders (Baldissera, Ganade, Brescovit, & Hartz, 2008), ants (Suguituru, Silva, Souza, Munhae, & Morini, 2011), beetles (Puker, Ad'Vincula, Korasaki, Ferreira, & Orozco, 2014), epiphytes (Boelter, Zartman, & Fonseca, 2011) and leaf litter taxa (Rocha et al., 2013). The impoverishment or even the absence of understory is crucial to loss of biodiversity in plantation forests. However, there are also "external" factors such as the absence of native habitat fragments near to the commercial tree stands which would be a source of individuals in the landscape using plantation forests as corridors or even to find food and shelter. In this paper, we addressed how both structure and configuration of the landscape affect species richness and community resilience of medium and large-bodied mammal species in plantation forests in southeastern Brazil. Our study aims to assess (1) which landscape attributes determine species richness and the persistence of mammals, and (2) what is the role of plantation forests for the conservation of mammals, mainly those species which are prone to local extinction due to habitat fragmentation.

2. Material and methods

2.1. Study sites

Our dataset combines our field data with data obtained in the literature (Dotta & Verdade, 2011; Mendonça, 2009; Silva, 2001; Silveira, 2005; Spínola, 2008). Field surveys were carried out in seven different plantation forest sites located in São Paulo State, Brazil (Fig. 1). We choose these sites because the field sampling was comprehensive and because they showed variability in the landscape structure surrounding the plantation forest sites. The size of plantation forest sites where the field surveys were carried out (not necessarily the area effectively surveyed) ranged from 88.99 ha (Passa Cinco) to 25,658 ha (Capão Bonito) with an average size of 9985 ± 10,660 ha (Table 1). The sites are between 30 and 205 km away of each other (Average Distance = 136 km).

In five of the sites, there were stands planted with *Eucalyptus* spp., *Corymbia* spp. and *Pinus* spp. tree species, whereas two of them (Pilar do Sul and Passa Cinco) had only *Eucalyptus* spp. and *Corymbia* spp stands. Although the commercial pool of species differed from one site to another, comparison of such local effects was beyond the scope of this paper. Natural vegetation surrounding these sites included Tropical Semideciduous forests, as well as ecotone transition to Cerrado and Dense Evergreen forest, the latter the typical vegetation found in Pilar do Sul site. The native vegetation cover was adjacent to plantation forests stands and/or interspersed between them. Within a 15 km radius from the center of them, Pilar do Sul had the highest percentage of native forest cover (49.2%) whereas the Floresta Estadual Edmundo Navarro de Andrade (hereafter FEENA) the lowest (8.2%; Table 1). Surroundings of FEENA also, had the smallest percentage of plantation forest cover (1.7%) and Capão Bonito had the highest (33.1%).

2.2. Data collection

Our field sampling at FEENA involved 98.3 km of census walked along three transects (3.5–5 km in length), following protocol described by Peres, Barlow and Haugaasen (2003), and 99.7 km of intensive

Table 1
Description of the studied plantation forests and surrounding landscapes in the São Paulo State, Brazil.

Site	Location	Size (ha)	Survey Methods	Vegetation Type	Annual Rainfall (mm)	% cover of landscape[a] Native	Plantation	Landsat image	Date	Reference
Pilar do Sul	22°54'48"S – 47°41'18"W	2250	Sand track-stations (2600)[b]	Semieciduous and Ombrophilous Forest	1683	49.23	9.32	ETM 220/076 and 220/077	26-Sep-1999	Silva (2001)
Itatinga	23°04'29"S – 48°36'59"W	2200	Sand track-stations (2160)[b]	Semideciduous Forest and Cerrado	1356	23.48	14.67	TM 220/076	30-Aug-2004	Silveira (2005)
Passa Cinco	22°21'59"S – 47°47'29"W	272	Intensive search (73.8)[c]	Semideciduous Forest and Cerrado	1534	16.02	6.59	TM 220/075 and 220/076	05-Sep-2006	Dotta and Verdade (2011)
Capão Bonito	23°55'15"S – 48°20'54"W	24921	Sand track-stations (3600)[b]	Semideciduous Forest and Cerrado	1407	17.78	33.17	TM 220/077	01-Aug-2005	Spínola (2008)
Agudos	22°26'35"S – 48°54'23"W	15523	Sand track-stations (750)[b] and Intensive search (303)[c]	Semideciduous Forest and Cerrado	1464	19.5	16.25	TM 221/075 and 221/076	21-Apr-2006	Mendonça (2009)
Lençois Paulista	22°49'17"S – 48°56'9"W	22417	Sand track-stations (750)[b] and Intensive search (346)[c]	Semideciduous Forest and Cerrado	1464	18.47	27.46	TM 221/076	21-Apr-2006	Mendonça (2009)
FEENA	22°24'23"S – 47°31'15"W	2314	Line transect (98.3)[c] and Intensive search (99.7)[c]	Semideciduous Forest and Cerrado	1534	8.26	1.77	TM 220/075 and 220/076	05-Sep-2006 and 19-Jul-2006	This study

[a] Landscape area is a 15 km radius circular buffer.
[b] sampling effort is given in trap-nights.
[c] sampling effort is given km walked.

searches for tracks along trails and firebreaks. All walks were conducted at 1.7 km/h average speed from 6:00–10:00 h and 14:00–18:00 h on days without rain between December 2005 and June 2007. Total sampling effort was 198 km resulting in 62 animal records (only two sightings). In the dataset compiled from literature review, the sampling protocols adopted used sand track-stations, intensive search for tracks, occasional records or a combination of these methods (Table 1). Sampling effort in all these areas was conducted for at least 6 months during a year and covered both plantation forest stands and adjacent native vegetation patches. In Table 1, we described the sampling effort carried out in plantation forest only. In three sites where sand track-stations was the main used method, the effort ranged from 2160 to 3600 trap-nights. When the main protocol for field sampling was the intensive search for tracks, the distance walked ranged from 73.8 to 346 km (Table 1).

We built a presence-absence matrix from data of species found in plantation forests stands or those seen in firebreaks and dirt roads between native vegetation and plantation stands, including small mammal species (> 1 kg; *Guerlinguetus aestuans* and *Cavia aperea*; Appendix A). For some species with uncertain identification, we grouped data at genus level (*Didelphis*, *Leopardus*, excluding *L. pardalis*, *Mazama*, *Guerlinguetus* and *Cavia*). Because of unreliable identification of porcupines at species level (*Coendou* spp. and *Sphigurus* spp. have nocturnal and arboreal habits), we excluded these data from the analysis. All species of our matrix have wide geographical distribution and are readily detected by the adopted sampling protocols. Thus, this allows us to compare the fauna in different plantation forest sites even when different sampling protocols and effort were carried out. We also gathered information on invasive species such as wild boar (*Sus scrofa*) and cape hare (*Lepus capensis*) that were recorded in some of the plantation forest sites (Appendix A). Therefore, our database consisted of 26 mammalian species and four genera as functional groups, providing a total of 30 taxa analyzed.

2.3. Functional diversity as a proxy of vulnerability to extinction

Human-induced changes in natural environments such as habitat loss, fragmentation and matrix roughness (i.e. how difficult it is to move across the landscape) affect species differently (Díaz et al., 2013; Tscharntke et al., 2012). Different responses to changes in the habitat have been related to some species-specific functional traits (Blaum, Mosner, Schwager, & Jeltsch, 2011; Hooper et al., 2005). According to Flynn et al. (2009), functional traits are measurable features of a given organism concerning its interaction with habitat and other organisms such as resources use, body growth rates, reproduction and survival. Certain functional traits are good indicators of species vulnerability to extinction in fragmented landscapes as they represent the ability of species to survive in modified habitats such as plantation forests.

In order to ensure that suitable functional traits were chosen, we review the literature of the main traits related to vulnerability of species to extinction and response to habitat changes (Table 2). Our proxy of species-specific vulnerability to extinction was based on nine functional traits provided by Pereira and Daily (2006) adapted for South American species. We added to this dataset the species feeding habitats and locomotion habitat categories provided by Paglia et al. (2012). Data on invasive species were removed from analysis of functional traits. We applied the Functional Diversity Indices (FD) proposed by Mason, Mouillot, Lee, and Wilson, 2005 and Villéger, Mason, and Mouillot, 2008 to represent the relative space and distribution of vulnerability to extinction of mammals present in plantation forests. We used the package FD (Laliberté & Legendre, 2010) available to R statistical environment version 3.1.2 (R Development Core Team, 2014) for the calculation of functional richness (*FRic*), functional evenness (*FEve*) and functional divergence (*FDiv*). In addition, we adjusted non-Euclidean portion of distance matrix using Cailliez correction eliminating negative eigenvalues (Legendre & Anderson, 1999).

The Functional Diversity Indices are suitable even when they are calculated from presence-absence data (Villéger et al., 2008). By definition, functional richness represents the amount of functional space occupied by community, functional evenness corresponds to the regularity of relative positioning of each species in the functional space and functional divergence represents the distance them from the center of functional space (Mouchet, Villéger, Mason, & Mouillot, 2010; Villéger et al., 2008). For our purposes, functional space could be designated as the range of species vulnerability to extinction in human modified landscapes. Thus, in functionally richer plantation forests, species vulnerability would be more variable. Plantation forests where functional evenness is high, the number and degree of vulnerable species would be equivalent to non-vulnerable species. In functionally divergent plantation forests, the degree of vulnerability between species would be more contrasting.

2.4. Landscape structure

We assessed the landscape structure adjacent to study sites using images of Landsat 5 Thematic Mapper and Landsat 7 Enhanced Thematic Mapper (30 m spatial resolution), acquired from the Brazilian National Spatial Research Institute (INPE; http://www.dgi.inpe.br/CDSR). We selected images obtained at the time corresponding to the field sampling of each study site (Table 1). In addition, when present, the area covered by clouds in these images was not large enough to preclude or compromise the quality of land cover classification.

We carried out the geometric correction of stacked non-thermal bands using the Third Order Polynomial algorithm and resampling method of Nearest Neighbor, available in software Erdas version 9.1 (Leica Geosystems, 2006). By mean of geometric correction, we generated at least 250 control points per image from reference Landsat images acquired from Global Land Cover Facility (http://glcfapp.glcf.umd.edu; acessed in 10 December of 2014), projected in UTM and Datum WGS-1984 zones 22 or 23 S. The global mean of RMS error for the geometric correction was 0.3161 ± 0.056 pixel.

We set a circular buffer with 15 km radius from the center of the plantation forest site surveyed. We adopted this arbitrary size to properly include in our analysis of landscape context, the whole area of these plantation forests and their surroundings. In order to perform the land-cover classification, we use the supervised method by mean of Maximum Likelihood algorithm only with the combination of bands 3, 4 and 5. We identified the major land cover classes present in the seven study landscapes previously to the classification procedure by collecting a minimum of 15 samples of spectral signatures for each of them. Thus, we produced maps classified as following: i) water; ii) natural habitats (forests and savanna "Cerrado" vegetation); iii) forest plantation (Eucalyptus and Pinus stands); iv) citrus plantation; and v) annual crops, pastures, urban areas and others uses, gathered as matrix. In order to improve the classification accuracy, we corrected misclassified areas based on high resolution images available in Google Earth® platform, recoding manually the misclassified pixels. We merged the land-cover classes citrus plantation and matrix for the completion of classified maps.

We calculated a suite of landscape metrics using Fragstats 4.2 (McGarigal, Cushman, & Enen, 2012) aiming to assess at how mammals respond to changes in the structure, spatial distribution and distance between native habitat patches and plantation forests stands. We used the suffix "nat" to designate the metrics related to the native habitat class and "euc" for plantation forest class. For each landscape, we calculated the percentage of area covered by each class (*PLAND*), the Fractal Dimension index (*FRAC_MN*) for quantifying the shape complexity of patches, Clumpiness index (*CLUMPY*), which represents the degree of aggregation of patches of same class based on pixel adjacencies and mean Euclidean Nearest Neighbor distance (*ENN_MN*). Moreover, we calculated the Interspersion and Juxtaposition index (*IJI*) at landscape level. *IJI* measures how immersed are patches of a given

Table 2
Functional traits for each mammalian species and functional importance to their survival in anthropogenic habitats.

Functional trait	Functional importance	References
Mean Body Size (g)	Larger bodied mammals are more susceptible to hunting and persecution. They also, have usually lower reproductive rate.	Purvis, Agapow, Gittleman, and Mace, 2000; Cardillo et al. (2005), Cardillo, Mace, Gittleman, & Purvis, 2006
Litter size	Species with small litters are less able to compensate for increased mortality.	Purvis, Agapow, et al. (2000); Cardillo (2003); Cardillo et al. (2006)
Litter interval	Longer litter interval is associated with lower reproductive rates.	Harcourt and Schwartz (2001); Jones, Purvis, and Gittleman, 2003
Breeding age (year)	Species with later sexual maturity are associated with lower reproductive rates.	Purvis, Gittleman, Cowlishaw, and Mace, 2000; Cardillo et al. (2006), 2008
Mean lifespan (year)	Longevity in mammals is inversely associated with potential reproductive rates.	Holliday (2005)
Estimated population growth rate in the native habitat (individuals/year)	Population growth rate mediates the extinction risk against environmental stochasticity and random catastrophes.	(Lande, 1993), Pereira and Daily (2006)
Critical patch diameter (km)	This is a measure of vulnerability to land use change whereby species with higher values require large home ranges.	Brashares (2003); Pereira and Daily (2006)
Feeding guild	The specific use of food resources is related to sensitivity to habitat fragmentation.	Vetter, Hansbauer, Végvári, and Storch, 2011; Blaum et al. (2011)
Foraging habit	The foraging habit represents the strategies and specific use of the habitat.	Blaum et al. (2011)

land-cover class between patches of other land-cover class present in that landscape. It ranges from 0, when the distribution of adjacencies of a given class tend to be uneven in regard to the other land-cover classes to 100, when all patches of all land-cover classes are equally adjacent.

2.5. Statistical analysis

In order to evaluate how the landscape context of plantations forests sites affects the species richness and functional diversity, we used redundancy analysis (RDA; Ter Braak & Prentice, 1988; Ter Braak 1994) having the landscape metrics as our explanatory variables. The first step of RDA is regress each response variable on all variables of the matrix of explanatory variables, and next, carry out a Principal Component Analysis from the matrix of fitted values generated by the regression obtaining eigenvalues and eigenvectors (Legendre & Legendre, 1998). We used the R environment (R Development Core Team, 2014) for all statistical analysis. First, we made a correlation matrix to verify the presence of correlation between the explanatory variables ($r > 0.75$). We found high correlation between *ENN_MNnat* and *IJI*, *CLUMPYnat* and both *CLUMPYeuc* and *PLANDnat*, *FRAC_MNeuc* and *PLANDeuc*, and *ENN_MNeuc* and *IJI*. Thereby, we opted to remove the variables *ENN_MNnat*, *CLUMPYnat*, *FRAC_MNeuc* and *ENN_MNeuc* from the analysis. Thus, to perform the redundancy analysis we used four response variables and five explanatory variables (Appendix B). Likewise in a regression analysis, RDA calculates a R^2 that explains the proportion of variation of response matrix by explanatory matrix. However, because the R^2 of RDA is biased, we also calculated the adjusted R^2 to evaluate the performance of explanatory variables (Peres-Neto, Legendre, Dray, & Borcard, 2006). Due to the potential influence of covariates, we controlled for the effects of collinearity by calculating the variance inflation factor of regression coefficients (Dormann et al., 2013). Moreover, we tested whether the relationship between the explanatory and response matrices (i.e. the global model), are different from random using a Monte Carlo test with 1000 permutations (Legendre, Oksanen, & ter Braak, 2011). We also tested the significance of relationships between canonical axes, and as well as of each explanatory variables.

3. Results

Seven of 30 species/functional groups were recorded in all sites, among them, habitat generalists such as nine-banded armadillo (*Dasypus novemcinctus*), crab-eating fox (*Cerdocyon thous*) and crab-eating raccoon (*Procyon cancrivorus*), large carnivores (puma; *Puma concolor*) and game species (brocket's deers *Mazama* spp.; see Appendix

A). Other native ungulate species were less common. Collared peccary (*Pecari tajacu*) in turn, were recorded in four areas, whereas lowland tapir (*Tapirus terrestris*) was found only in two sites (Pilar do Sul and Capão Bonito). There were no records of primate or white-lipped peccary (*Tayassu pecari*) in all of seven plantation forest sites sampled. The invasive species were recorded in four plantation forest sites (Itatinga, Passa Cinco, Agudos and FEENA). *L. capensis* was recorded in four areas whereas *S. scrofa* was recorded in Passa Cinco and FEENA, the two sites with the lower proportion of landscapes covered by native vegetation.

The first two RDA axes explained 92.6% of variance of fitted data (58.8 and 33.7%, respectively). There was little difference between RDA R^2 (0.987) and the adjusted RDA R^2 (0.924), indicating that the strong relationship between explanatory and response matrix found was properly estimated. The test of significance for the global model with 1000 permutations corroborated this result ($F_{3.94} = 15.648$, $df = 5$, $p = 0.028$), as well as the test for the first canonical axis ($F_{2.35} = 93.302$, $df = 1$, $p = 0.021$) and the second one ($F_{1.35} = 53.522$, $df = 1$, $p = 0.019$). Similarly, the test for each explanatory variables showed highly significant results for *PLANDnat* and *CLUMPYeuc* (Table 3). The regression coefficients did not exhibit the influence of covariates indicating the absence of collinearity between our explanatory variables, once all variance inflation factors were less than 10 (Dormann et al. 2013; Table 3).

The explanatory variables that showed significant relationships with matrix of response variables had opposite results (Fig. 2). The proportion of landscape covered by native habitats (*PLANDnat*) was positively correlated with functional richness (*FRic*), whereas the degree of aggregation of commercial tree stands (*CLUMPYeuc*) was negatively correlated with *FRic*. Indeed, more native vegetation in the landscape and less aggregation of plantation forests stands contribute positively to the

Table 3
Significance test of explanatory variables constrained on the RDA axes and the variance inflation factors (VIF) to control the collinearity.

Explanatory variable	Variance	F value	p	VIF
PLANDnat	1.5418	30.54	0.008	7.795
FRAC_MNnat	0.4691	9.293	0.042	3.253
PLANDeuc	0.5498	10.89	0.035	8.162
CLUMPYeuc	1.1417	22.61	0.018	4.538
IJI	0.2470	4.894	0.115	6.835

PLAND = Proportion of landscape; *FRAC_MN* = Fractal Dimension index; *CLUMPY* = Clumpiness index; *IJI* = Interpersion and Juxtaposition index.
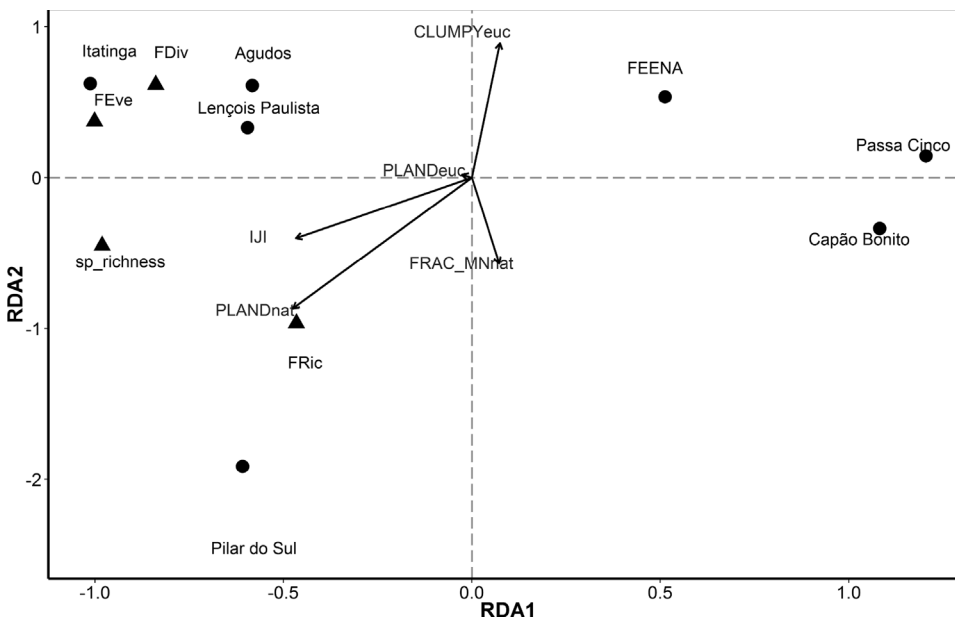
Fig. 2. RDA triplot of correlations of variables and plantation forest sites constrained by landscape metrics eigenvalues. Circles indicate the seven plantation forest sites, and triangles and arrows represent response and explanatory variables, respectively.

increasing of functional richness. On the other hand, species richness is not strongly correlated with *PLANDnat*, although the relationship tends to be positive. Functional Evenness (*FEve*) and Functional Divergence (*FDiv*) did not show any clear relationship with the significant explanatory variables.

4. Discussion

The effectiveness of plantation forests for conservation is strongly dependent on the landscape context, mainly regarding the structure and spatial distribution of native habitat fragments throughout the landscape. Our results suggest that the area of the landscape covered by natural habitats is critical to ensure that species more vulnerable to extinction can persist in the landscape and occur in plantation forest. Landscape context appears to be important not only in *Eucalyptus* and *Pinus* plantations. Alongside oil palm plantations in South-east Asia, the variation on species richness is strongly explained by total area of native habitats (Azhar, Lindenmayer, Wood, Fischer, & Zakaria, 2014). Moreover, distance from oil palm plantations to native habitat fragments has strong negative relationship with species richness, where threatened species such as Malayan tigers, sun bears and Asiatic tapir were recorded only within 5 km of forest fragments (Azhar et al., 2014; Yue, Brodie, Zipkin, & Bernard, 2015). In rubber plantations also, the distance and the area of native habitats in the landscape are positively related to overall bird species richness and forest dependent species (Zhang, Chang, & Quan, 2017).

The combination of spatial aggregation of commercial trees stands and low cover of native habitats in the landscape might lead to impoverishment of the community of medium and large-bodied mammals found in plantation forests. In spite of the relationship between proportion of native vegetation cover and functional diversity, this study did not assess the conditions of those native habitat fragments, and this variable could be considered in further studies since native vegetation fragments usually present high variation of conditions resulting in different habitat quality (Ferraz et al., 2014). On the other hand, plantation forests managed by pulp and fiber industries keep adjacent areas for conservation with native vegetation, as requirements of the Brazilian Forest Code and forestry management certification (Brockerhoff et al., 2013). These set aside areas can improve the conservation value of plantation forests, once commercial tree stands only has a limited role in biodiversity conservation because the mammalian fauna found there is often similar to or more impoverished than that found in small

and isolated fragments of Atlantic forest (Beca et al., 2017; Canale, Peres, Guidorizzi, Gatto, & Kierulff, 2012; Chiarello, 1999; Dotta & Verdade, 2011).

In a literature review, Ramírez & Simonetti (2011) found species richness and abundance of mammals in plantation forests were always lower than found in native forests. For example, tapirs were rare or even absent from plantation forests, whereas white-lipped peccaries were not found in any of the plantation forest sites studied here. These species are known to be more vulnerable to extinction due to habitat loss and fragmentation (Jorge, Galetti, Ribeiro, & Ferraz, 2013) as well as hunting pressure (Cullen Jr. & Bodmer, 2001). Thus, their presence in plantation forests could be strictly dependent on surrounding native habitat. Others studies have found the same pattern for multiple taxa comparing species richness between commercial tree stands and fragments of native vegetation. For example, the species richness of birds showed marked impoverishment in *Eucalyptus* and *Pinus* plantations (Barlow, Mestre, Barlow, Mestre, Gardner, & Peres, 2007; Marsden et al., 2001; Volpato et al., 2010; Zurita et al., 2006), as well as small mammals (Martin et al., 2012), Cetoniinae beetles (Puker et al., 2014), lizards, myriapods, arachnids (Fonseca et al., 2009; Rocha et al. 2013), ants (Rocha et al., 2013; Suguituru et al., 2011), vascular epiphytes (Boelter et al., 2011; Fonseca et al., 2009), and fungi, galling insects, butterflies, and flatworms (Fonseca et al., 2009). However, when compared with other human land-uses, plantation forests definitely hold more species of medium and large mammals than pastures and annual crops. Even the plantation forest site with the lower species richness (Passa Cinco; $S = 15$) hold more species than neighbor pastures ($S = 10$) and neighbor sugarcane crops ($S = 13$) (Dotta & Verdade, 2011).

At the plantation forest scale, the impoverishment of mammal community, particularly of arboreal species, could be also attributed to larger canopy openness as well as changes in understory such as lower diversity, lower basal area (Barlow, Gardner, et al., 2007; Boelter et al., 2011; Zurita et al., 2006) and the lack of plants with fleshy fruits and epiphytes (Fonseca et al., 2009). The absence of primate species in all plantation forest sites studied is the most noticeable consequence of habitat simplification (Coelho, Juen, & Mendes-Oliveira, 2014; Stallings, 1991). Primate groups were recorded in the forest fragments adjacent to commercial tree stands in Pilar do Sul, Agudos and Lençois Paulista (Mendonça, 2009, Silva, 2001). Groups of black-horned capuchin monkeys (*Sapajus nigritus*), brown howler monkeys (*Alouatta guariba*), black-pencilled marmosets (*Callithrix penicillata*) and even the

endangered black-lion tamarins (*Leontopithecus chrysopygus*) were observed foraging in the border of fragments, but they were not seen crossing the firebreaks towards commercial tree stands. In contrast, capuchin monkeys were observed damaging trees of Pine stands in months of scarce availability of fruits in native habitat fragments in Southern Brazil (Mikich & Liebsch, 2014). Moreover, flowers, seeds and nectar of the *Eucalyptus* trees were recorded as food items by bearded capuchin monkeys (*Sapajus libidinosus*; Freitas, Setz, Araújo, & Gobbi, 2008).

Primate species are able to persist in the plantation forests but not exclusively on it and only under certain conditions. For example, howler monkeys groups living in *Eucalyptus* plantation with unmanaged understory had similar mean group size from those found in continuous forest (Bonilla-Sánchez, Serio-Silva, Pozo-Montuy, & Chapman, 2012). Although, howler monkeys were often seen foraging in the understory, they spent the most of time feeding in adjacent native forest fragments (Bonilla-Sánchez, Serio-Silva, Pozo-Montuy, & Chapman, 2012).

The presence of invasive species in the plantation forests sites reveals a major concern for conservation management, since wild boars and cape hares are spreading across the Brazilian territory (Auricchio & Olmos, 1999; Pedrosa, Salerno, Padilha, & Galetti, 2015). Moreover, the temporal range of field surveys synthesized here (~7 years) may give an underestimation about the presence of invasive species on plantation forest sites. Although it requires corroboration, is plausible the hypothesis in which plantation forests located at highly fragmented landscapes suit as refuge for invasive such as wild boars and cape hares.

Usually, the trade-offs between land-sparing and land-sharing management have been applied for areas of food production (Fischer et al. 2014). In plantation forests, gains in biodiversity and profitable harvest would be better achieved with management intensification of commercial tree stands, protection and restoration of surrounding native habitats, and improvement of landscape connectivity. For example, logging operations in Borneo achieved higher levels of biodiversity whether they combine areas with high-intensity harvest and sparing areas without timber extraction (Edwards et al., 2014). This wildlife-friendly management finds legal support in the Brazilian forest code (Galetti et al., 2010) and should be reinforced by certification mechanisms (Brockerhoff et al., 2013). In terms of fulfillment of legislation and management certification, the pulp and fiber sector in Brazil is more willing to accomplish these requirements than others sectors of economy, such as soybean and cattle (Azevedo et al., 2017; Martinelli, Joly, Nobre, & Sparovek, 2010).

These results highlight the importance of landscape context and the conservation value of small and isolated native habitat fragments for medium and large mammals. The value for conservation of plantation forests is necessarily linked with the conservation of native habitat fragments in their midst. In landscapes where natural habitats are highly reduced and fragmented, the importance of plantation forest cannot be depreciated. For those species that persist in these landscapes, plantation forests can facilitate movement and dispersion between fragments improving the permeability of landscape, mainly compared with open environments like pastures and annual croplands. In the case of the Brazilian Atlantic forest, more permeable landscape can be crucial for the viability of many populations of medium and large-bodied mammals.

Acknowledgements

Appendix A. Supplementary data

Supplementary material related to this article can be found, in the online version, at doi:https://doi.org/10.1016/j.jnc.2017.11.009

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