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Landscape structural analysis of the Lençóis Maranhenses national park: implications for conservation

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ABSTRACT

Our work evaluated the anthropic effects on the landscape structure of the Lençóis Maranhenses National Park (LMNP) and its Buffer Zone, and proposed strategies for the region's conservation. LMNP is an important protected area in Brazilian north coast which protects a unique wetland ecosystem composed of sand dunes fields and a coastal vegetation called *restinga*. Supervised mapping of LMNP and a surrounding buffer of 3 km was carried out through high resolution and fine scale (1:5000) satellite images. The mapped area was subdivided in 1000 ha hexagonal Analysis Units (AU) and the following landscape metrics were calculated for each one of them: cover area (CA) of each soil cover class - dune fields (CA-DUNES), water bodies (CA-WATER), dense restinga (CADENSE), scattered restinga (CA-SCATTER), grassland (CA-SANDY), mangroves (CA-MANG), anthropogenic activity (CA-ANTRO) and, secondary vegetation (CA-SECOND); Landscape Shannon Diversity Index (SHDI), and; percentage of native vegetation cover (NV − COV). Pearson correlations were performed between the CA of each class and SHDI to identify the classes most correlated to CA-ANTRO. Our results showed that anthropic classes (crops, trails, and villages) had a stronger correlation (Pearson Correlation, $r \approx 0.65$) with phytophysiognomies of dense *restinga*, secondary vegetation and SHDI, thus indicating that the land use conversion occurs in dense *restinga* areas and promotes vegetation secondarization, as well as increasing fragmentation. At least, 42% of the dense *restinga* habitats was destroyed due to human activities. Five conservation and restoration strategies were proposed in a local scale depending on the percentage of native vegetation cover on each AU, from the most to less conserved: (a) only conservation; (b) conservation with management; (c) management; (d) management and restoration; and, (e) restoration. The implementation of Agroforestry Systems with agro-successional restoration goals was recommended as an alternative for land use.

1. Introduction

Land use conversion for anthropic activities is the main threat for large-scale terrestrial biodiversity conservation on the planet, as it reduces the amount of available habitat to organisms, as well as causing the fragmentation of remaining habitat, creating patches of suitable habitat surrounded by a matrix of lower conservation value (Fahrig, 2017; Turner, 2005). Estimates indicate that when only 30% of the landscape habitat remains (fragmentation threshold), species survival probability is drastically reduced (Fahrig, 2001; Martensen, Ribeiro, Banks-Leite, Prado, & Metzger, 2012). To increase organism persistence in the landscape, nature conservation depends on the maintenance of conserved habitat parcels and ecological corridor networks connecting them, thus guaranteeing suitable habitat and enabling organism dispersal in the landscape (Haila, 2002).

Fragmented landscapes are usually represented by the "fragmentation model", where habitat fragments are compared to islands, and the matrix to the oceans (Haila, 2002). McIntyre and Hobbs (1999) have proposed the "*variegation* model" where habitat loss and fragmentation increase the diversity of coverage classes. Areas of destroyed habitat

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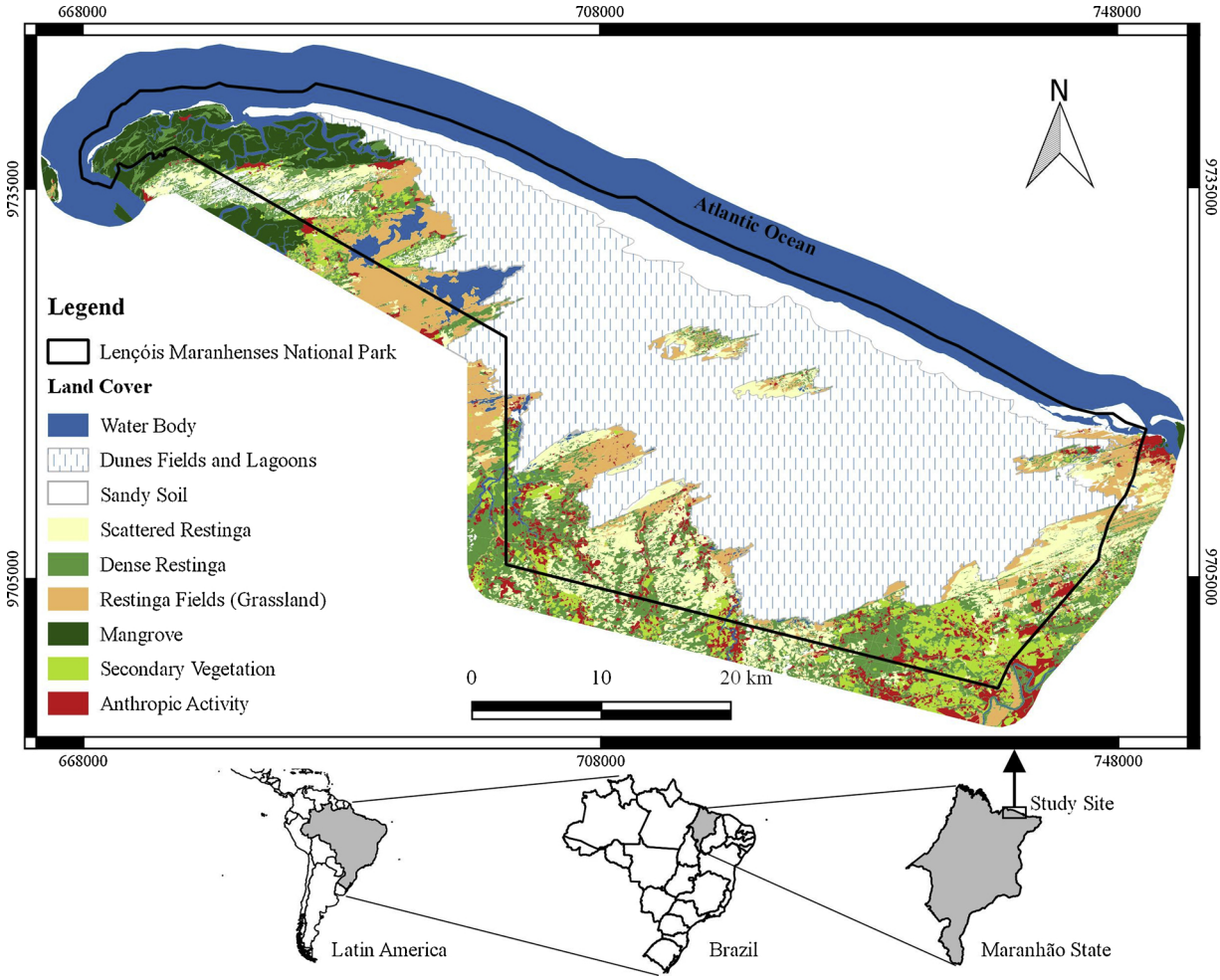

Fig. 1. Location and land cover types of the study area: Lençóis Maranhenses National Park, Maranhão state, northeastern Brazil.

will have different uses, such as settlements, roads, crops and regenerated areas and organisms will respond differently to each of these patches.

Over the last 30 years, landscape ecology has developed dozens of metrics and indices to interpret landscape spatial settings and its influence on biodiversity. However, its real ecological meaning or applicability in conservation programs is still debatable (Corry & Nassauer, 2005; Kupfer, 2012). Recent studies have demonstrated that metrics for land cover classes are more effective to identify organism distribution patterns because they provide information on the amount of available habitat to organisms in a given landscape (Fahrig, 2017) and present clearer and more parsimonious answers to the variation on biodiversity indices (Cunningham & Johnson, 2011; Lustig, Stouffer, Roigé, & Worner, 2015). Hence, the analysis on how land use conversion affects the native vegetation cover distribution is essential for territorial planning focused on conservation (Barreto et al., 2010; Gangadharan, Vaidyanathan, & St. Clair, 2017; Villard & Metzger, 2014).

The management and establishment of protected areas, formal or not, is one of the main *in situ* conservation tools used for biodiversity maintenance around the globe, precisely due to the protection of extensive areas from human impacts (Herrmann, Machado, & Macedo, 2011). Protected areas have acted as an efficient tool for biodiversity conservation and structural and functional maintenance of ecosystems on a large scale. However, bad planning and inadequate management can compromise its efficiency and put at risk the conservation efforts (Joppa & Pfaff, 2011; Pfaff, Robalino, Herrera, & Sandoval, 2015; Rouget, Richardson, & Cowling, 2003; Santos & Schiavetti, 2014). Landscape resilience depends on adequate planning, the flexibility of environmental management institutions and the development of solutions for local-scale problems through deliberate experimentation with participation and social learning (Cumming, Olsson, Chapin, & Holling, 2013).

Lençóis Maranhenses National Park (LMNP) is a coveted tourism destination in northeastern Brazil and shelters important coastal ecosystems of sand dunes, coastal vegetation called *restingas*, and mangroves (BRASIL, 1981). About 1160 families of small landowners live distributed in over 40 villages inside and arround the LMNP. This population develops subsistence agriculture through the conversion of native vegetation into intermittent cultivation, commonly using fire. Fishing, hunting, grazing, and collecting wood and fruits are also common activities in the LMNP (Abakerli, 2001; IBAMA, 2002; IBGE, 2017). Even though foreseen by law (BRASIL 2000, 2002), currently no action relating to land ownership regularization or establishment of Commitment Terms with the families was taken to reduce anthropic pressures over LNMP's resources. In addition to the land issue, the LMNP is also threatened by the growth of the tourism industry, especially with regard to motorized visitor traffic and real estate speculation.

For a better understanding of the interaction between anthropic activities and vegetation, we propose a conceptual model for land use for the region. It foresees that the areas most impacted by agriculture will be concentrated in depressions and throughout hydric bodies where denser and resource-richer vegetation develops due to the greater water and nutrient availability in the soil. Plateau, mangrove, and field areas are less prone to conversion to anthropic activities since

Table 1
Results of cover area (CA) in the broad and restricted class maps.

Broad classes	Area (ha)	%	Restricted classes	Area (ha)	% of broad class	% of total
Dunes fields and lagoons CA-DUNES	84,514.93	38.08	Dunes fields and lagoons	84,514.93	100	38.08
Water body CA-WATER	46,263.00	20.85	Ocean	40,509.62	87.56	18.25
			River	3,030.12	6.54	1.36
			Lake	2,723.27	5.88	1.22
Dense restinga CA-DENSE	23,825.48	10.74	Arboreous	11,847.72	49.72	5.33
			Shrubs	8,985.33	37.31	4.04
			Riparian forest	1,439.32	6.04	0.64
			Mix of arboreous and shrubs	926.66	3.88	0.41
			Mix of arboreous, shrubs and field	306.81	1.28	0.13
			Palm tree forest	200.28	0.84	0.09
			Spread arboreous	99.42	0.41	0.04
			Litoral restinga	19.89	0.08	0.004
Restinga fields CA-GRASS	14,481.40	6.52	Open field (grassland)	5792.13	39.99	2.61
			Mix of field and scattered shrubs	4946.60	34.15	2.22
			Lakeside grasslands	3,406.24	23.52	1.53
			Meadow	317.56	2.19	0.14
			Mix of field and palm tree	18.82	0.001	0.003
Scattered restinga shrubs CA-SCATTER	14,910.90	6.73	Scattered restinga shrubs	14,910.90	100	6.73
Sandy soil CA-SANDY	12,098.20	5.45	Beach	10,210.50	84.39	4.60
			Sandy soil	1,772.00	14.64	0.79
			Dune	67.72	0.55	0.03
			Sand bank	47.92	0.39	0.02
Mangrove CA-MANG	7,815.45	3.52	Mangroove	6,764.29	86.54	3.04
			Apicum	1,051.16	13.44	0.47
Antrhopic activity CA-ANTRO	8,651.35	3.89	Crops	5622.00	64.98	2.53
			Trails	1,141.68	13.19	0.512
			Village	745.43	8.72	0.33
			Pasture	419.13	4.84	0.18
			Mix of pasture and scatter shrubs	290.77	3.36	0.13
			Urban zone	242.52	2.80	0.10
			Exposed soil	189.28	2.18	0.08
Secondary vegetation CA-SENCOND	9,158.19	4.13	Intermediary secondary vegetation	4,264.44	46.56	1.92
			Late secondary vegetation	4,003.13	43.71	1.80
			Pioneer secondary vegetation	890.61	9.72	0.40
Cloud cover	163.99	0.07	Cloud cover	163.99	100	0.07
Total	221,882.88	100		221,882.88		100

they are inadequate for agriculture.

Analysis of the impacts caused by the traditional land use in the LNMP and its buffer zone are imperative to guide administrative actions at the local scale by managers and decision-makers aiming to propose adequate conservation strategies for the region. This study aims to: (i) test the conceptual model for land use and occupation in the region; (ii) evaluate the effect of land use conversion over the landscape structure of LNMP and its surroundings; and, (iii) propose conservation strategies for the region.

2. Methods

2.1. Study area

Lençóis Maranhenses National Park (LMNP) is located at the eastern coast of Maranhão state, northeastern Brazil, between the coordinates 02°19′ and 2°45′ S; 42°44′ and 43°29′ W (Fig. 1) in an ecotonal area between the Amazon, Caatinga, and Cerrado biomes. It comprises an area of 155,000 ha partially overlaid by Barreirinhas, Santo Amaro do Maranhão, and Primeira Cruz municipalities. Climate is of Equatorial Zone with two marked seasons throughout the year. The rainy season extends from March to June and the dry season covers the rest of the year. Annual precipitation is 1800 mm and mean temperature is of 28.5 °C with a 1.5 °C amplitude (IBAMA, 2002). Soils are sandy and rich in quartz of Quaternary origin. Marine transgression and regression events occurring since the Pleistocene deposited sediments on the continent coast and formed successive dune fields during the last 100,000 years (Parteli, Schwaemmle, Herrmann, Monteiro, & Maia, 2006). The currently active vegetation-free dune field develops over inactive paleo-dunes from the Pleistocene, which are stabilized by a coastal vegetation (Herrmann, Andrade, Schatz, Sauermann, & Parteli, 2005; Luna, de, Parteli, & Herrmann, 2012).

Restingas are pioneer coastal vegetation formations that develop on sandy soils with fluviomarine influence and present different phyto-physiognomies related to edaphic features (Santos-Filho, Almeida, & Zickel, 2013). In the Pleistocene paleo-dune plateaus, the sclerophytic shrub *restinga* vegetation occurs in low densities, as scattered bushes, due to low soil humidity and nutrient availability, with occurrence of *Humiria balsamifera* and *Byrsonima sp*. On valley slopes, the sclerophytic woody *restinga* vegetation develops homogenously-distributed with closed canopy, but varying between shrubs and trees, with occurrence of *Hymenaea parvifolia*, *Anacardium microcarpum,* and *Cereus jamacuru. Mauritia flexuosa, Euterpe oleacea* and *Scripus sp,* are common throughout the water courses. On the margins of rivers, lakes, and lowland areas, that may or may not be subject to flooding in the rainy season, herbaceous *restinga* fields develop, which can be interspersed by shrubs and scattered trees (Santos-Filho, Almeida, Soares, dos, & Zickel, 2011). In these grasslands are found *Cyperus* sp., *Cassia rotundifolia*, *Borreria* sp., *Copernica prunifera* and *Astrocaryum vulgare*. In the coastal flooding plains typical mangrove forests of the Equatorial America develop, with the occurrence of *Rhizophora mangle*, *Avicennia schaueriana* and *Laguncularia racemosa*.

2.2. Mapping

Mapping of the studied area for the landscape structure analysis was performed with the Quantum GIS V.2.14.14 software (QGIS DEVELOPMEN TEAM, 2016) using the OpenLayer complement which provides

high resolution (4800 × 3021 pixels) satellite images from Google Earth (Steiniger & Hay, 2009; Zaragozí et al., 2011). Mapping was conducted between February and March of 2017, and during that period the most recent satellite images available on Google Earth were from December 2013 and June and August 2016 Google (2017). The mapped area was defined by the addition of a 3 km buffer over the shape file of the LMNP limits (ICMBio, 2018), thus encompassing the protected area and 3 km around it. The mapping scale used was of 1:5,000, since it is a highly heterogeneous environment, common to coastal areas and the fine scale of impacts over the landscape (Tomaselli, Tenerelli, & Sciandrello, 2012). Mapping was carried out through supervised classification through the creation of polygons in the internal buffer area delimiting the different landscape features that were classified according to land cover classes. Field visits were conducted to validate satellite images and identify the different vegetation cover physiognomies. Secondary vegetation was identified by verifying satellite images of previous dates in search for evidence of past anthropic interferences (for example: intermittent farming). When there was not enough evidence that a given fragment consisted of secondary vegetation, this was classified as native vegetation.

The mapping of the studied area identified 34 land cover classes that were grouped into broader classes to facilitate the statistical analysis (Table 1). The final mapping result were two shapefile archives, one with broader classes and the other with the specific cover classes, containing polygons that represent vegetation fragments and further soil cover classes. After mapping, the shape archives were converted to raster files with a 10 m resolution for landscape analysis.

2.3. Landscape structure analysis

The mapped area was subdivided into identical hexagonal Analysis Units (AU) with 1000 ha each for the landscape metrics analysis and, therefore, for landscape structure analysis. The use of hexagonal sampling units has been indicated for the investigation of landscape structure (Birch, Oom, & Beecham, 2007), already being successfully applied in Cerrado (savanas) areas in Maranhão (Barreto et al., 2010). AUs of larger scales (3000 ha) were generated for each initial AU to represent the regional scale. However, preliminary exploratory analyses indicated high Pearson correlation among these two scales ($r > 0.90$, $p < 2.2^{-16}$), thus the local scale was the only one used.

Metrics were calculated by the FRAGSTATS software (McGarigal & Marks, 1995). For each AU the following metrics were calculated: total cover area (CA) for each class – sand dune field (CA-DUNES), water body (CA-WATER), dense *restinga* (CA-DENSE), grassland (CA-GRASS), scattered *restinga* (CA-SCATTER), sandy soil (CA-SANDY), mangroves (CA-MANG), anthropic activity (CA-ANTRO), and secondary vegetation (CA-SECOND); native vegetation cover (NV − COV), and Landscape Shannon Diversity Index (SHDI). Class metrics were calculated using the broad classes map, while SHDI was calculated at the landscape level using the specific cover classes map. This was done because it is a landscape metric and different cover classes (anthropic or natural) influence its value (Jurasinski & Beierkuhnlein, 2006; Neel, McGarigal, & Cushman, 2004). Equation details for metric calculations can be obtained in McGarigal and Marks (1995).

The NV − COV metric was obtained by adding the area of native vegetation classes (CA-DENSE, CA-SCATTER, CA-GRASS and CA-MANG) divided by AU total area and multiplied by 100, thus obtaining the percentage of occupied area per native vegetation class in each hexagon (AU). Metrics of vegetation cover area have shown to be adequate to determine the conservation status and species richness patterns of a determined landscape (Cunningham & Johnson, 2011; Lustig et al., 2015).

SHDI metric measures the diversity of landscape fragments, being equal to 0 when there is only one fragment class in the landscape and increasing along with the amount of different fragment classes or the proportional fragment distribution of distinct classes. This metric has been demonstrated to be adequate to determine the landscape texture (heterogeneity), and is positively related to biodiversity (Concepción, Diaz, & Baquero, 2008). SHDI also indicates native vegetation fragmentation due to the increase in landscape texture (Griffith, Martinko, & Price, 2000).

Pearson correlations were calculated using the metrics results in AU to verify correlations between them, in order to analyze the effects of human activities over the landscape structure. Our hypothesis is that CA-ANTRO will be more correlated ($r \approx 0.70$) with CA-DENSE, CA-SECOND and SHDI than with other metrics. All calculations were performed in the R version 3.3.1 software (R Development Core Team, 2011).

2.4. Conservation strategies

We proposed conservation strategies based on NV − COV and have the goal of maintaining conserved habitat fragments and increasing landscape connectivity through matrix enrichment, according to Barreto et al. (2010) adaptation: only conservation (CS − CONS); conservation with management (CS − CONS-MANAG); management (CS-MANAG); management and restoration (CS-MANAG-RES); and, restoration (CS-RES). AU with high NV − COV values are more conserved, requiring less handling and restoration activity. As the value of NV − COV decreases, management and restoration activities gradually become more important.

3. Results

3.1. Landscape structure analysis

Total mapped area was 221,882.88 ha with dune fields (38%) and water bodies (21%) occupying 59% of it. The most abundant vegetation cover class was dense *restinga* (CA-DENSE) (11%) followed by scattered *restinga* (CA-SCATTER) and grasslands (CA-GRASS), both with 7%, and mangroves (CA-MANG) with 4%. Arboreous *restinga* patches represented almost half of the CA-DENSE class, followed by shrub formations (37.31%). Anthropic activities (CA-ANTRO) and secondary vegetation (CA-SECOND) areas each represented 4% of the mapped area. Farming areas and trails contributed the most to CA-ANTRO. The most frequent secondary vegetation areas were of intermediate and late ecological succession stages. There was little cloud coverage in the study area (0.07%), therefore, the analysis was not affected. The summary of soil cover percentages is in Table 1 and the final map (Fig. 1).

Pearson correlation results between the different cover classes supported the conceptual land use model proposed (Table A1). Anthropic activities are more correlated with dense *restinga* areas ($r = 0.66, p < 2.2^{16}$) than with scattered *restinga* areas ($r = 0.32, p = 1.59^{-07}$), grasslands ($r = 0.14, p = 0.03$), and mangroves ($r = -0.04, p = 0.416$). Anthropic activities are also moderately correlated with the secondary vegetation ($r = 0.67, p < 2.2^{16}$), which is the result of the crop rotation system. SHDI yielded a moderately positive correlation with the anthropic classes and dense *restinga* (0.66 and 0.65, respectively, and both with $p < 2.2^{-16}$).

The analysis of native vegetation cover (NV − COV) showed that 22.34% of Analysis Units (AU) represented 0–40% of the native vegetation, *i.e.* there are more areas converted to anthropic use and secondary vegetation than conserved native habitat. On the other hand, 26.43% of the AU represented 60–100% native vegetation cover, thus considered as more preserved areas. Almost half of the AU (41.38%), which correspond to areas naturally without vegetation, such as the ocean or the active dune field, did not present vegetation cover, as expected (Table A2 and Fig. 2A).

3.2. Conservation strategies

Fig. 2-A shows the result of the conservation strategies for each AU.

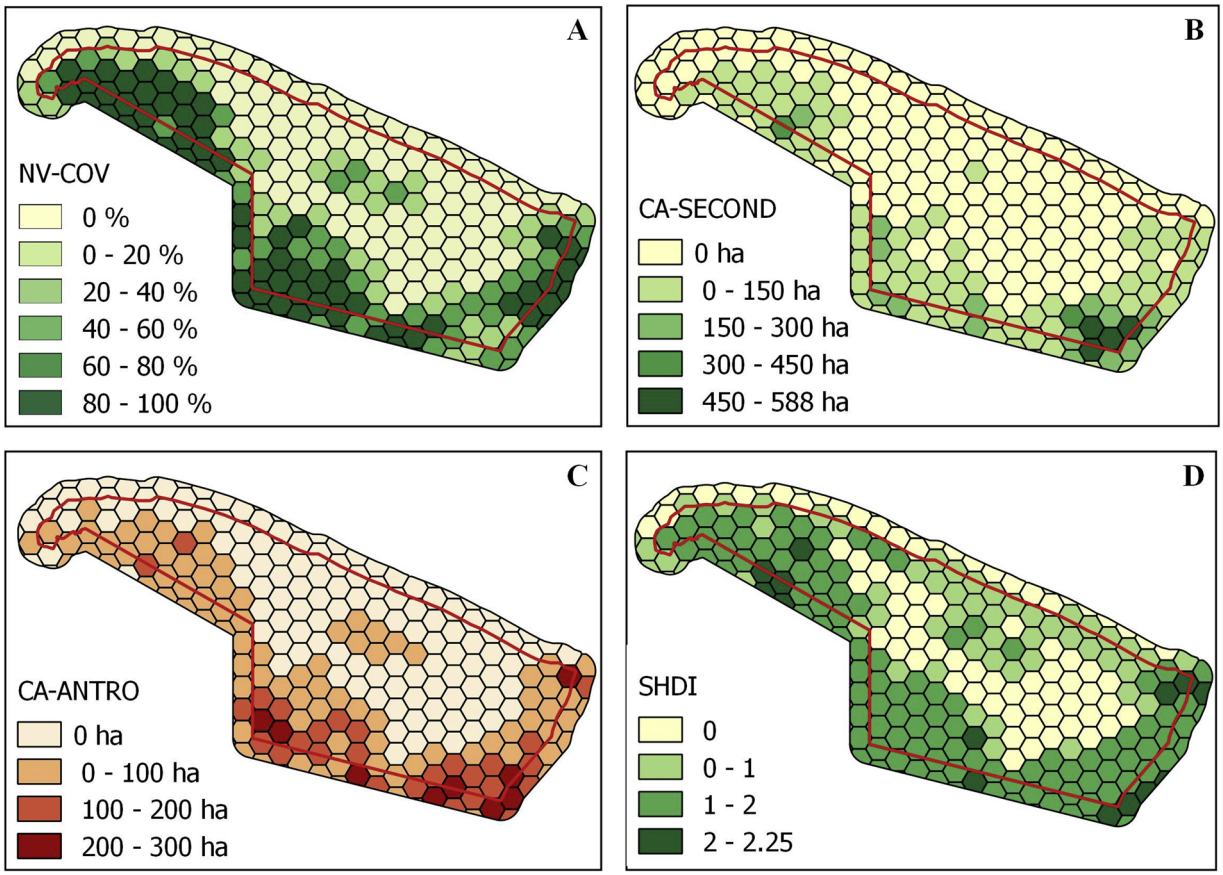

Fig. 2. AU metrics results. A- Percentage of Native Vegetation Cover (NV – COV). B – Secondary Vegetation Cover Area (CA-SECOND). C – Anthropic Activity Cover Area (CA-ANTRO). D – Shannon Diversity Index (SHDI). AU with results equal to 0 correspond to areas occupied by dune fields and the ocean. LMNP limits are shown by the red line (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article).

Table 2
Summary of actions to be developed in the AU.

Strategies	NV-COV	Land regularization	Land use
CS-CONS	80-100 %	Very urgent	Conservation
CS-CONS-MANAG	60-80 %	Urgent	Conservations and managment
CS-MANAG	40-60 %	Short-term	Managment
CS-MANAG-RES	20-40%	Midterm	Managment and agro-sucessional restoration
CS-RES	0-20%	Long-term	Agro-sucessional restoration

Actions and priorities were elaborated considering the current Brazilian environmental legislation (BRASIL 2000, 2002) in order to enable its real *in situ* application (Table 2).

according to NV – COV results.

- CS – CONS – *AU* are well-conserved (80–100% of habitat), presenting low anthropic pressure. Land regularization activities focused on conservation must be developed in these *AU*s to avoid new land use conversions and native vegetation loss. The implementation of agricultural activities and village expansions must be discouraged in these areas.
- CS – CONS-MANAG – *AU*s are moderately preserved (60–80% of habitat), presenting more native vegetation fragments than anthropic or secondary areas. In these *AU*s, land regularization actions focused on conservation should be conducted together with landscape management. Implementation of permanent agricultural activities and village expansion should be discouraged in these areas.
- CS-MANAG – *AU*s are in an intermediate conservation state (40–60% of habitat), therefore actions focused on habitat fragments maintenance and matrix quality improvement should be prioritized. Agroforest Systems (SAF) can be implemented in areas already converted and in secondary-vegetation fragments with the final goal of short-term agro-successional restoration.
- CS-MANAG-RES – *AU*s are highly affected by land use conversion (20–40% of the habitat) presenting more secondary vegetation areas and anthropic activities than native vegetation. Resource application for conservation in these areas will be a small contribution to biodiversity conservation. Agroforestry Systems must be implemented in the secondary vegetation fragments in ecological succession stages with the final goal of midterm agro-successional restoration.
- CS-RES – For *AU*s in critical situation (0–20% of the habitat), resource application towards conservation will be a very small contribution to biodiversity conservation. Therefore, these areas could be used for crop productions that are less aggressive to biodiversity. Agroforestry Systems should be implemented in these already-converted areas and in the secondary vegetation fragments with the final goal of long-term agro-successional restoration.

4. Discussion

4.1. Landscape structure analysis

Our results demonstrate that the traditional farming practices of *Lençóis Maranhenses* communities are capable of altering the LMNP landscape structure and its surroundings, but still maintains valuable habitat fragments for conservation. This setting resembles the "*variegation* model" more than the "fragmentation model" according to McIntyre and Hobbs (1999), as there was no transformation of a homogenous landscape into a binary one (fragment x matrix). In fact, an increase in diversity of cover classes represented by a complex matrix composed of crop and trail areas, as well as secondary vegetation in different succession stages and native vegetation fragments of distinct classes occurred (Fischer, Lindenmayer, & Fazey, 2004).

In our study, we observed that anthropic activities occur throughout the whole area occupied by *restinga* vegetation (Fig. 2C), thus evidencing that trails, villages, and crops occur near natural environments. The proximity of human activities to natural environments and the moderate and chronic land use promote biological and phylogenetic impoverishment of Caatinga's vegetation (2016, Ribeiro, Arroyo-Rodríguez, Santos, Tabarelli, & Leal, 2015). Moderate and chronic use of natural resources alter the Indian savannah forest's functional composition, thus harming native species (Mandle & Ticktin, 2015).

Crop areas were the larger contribution to CA-ANTRO and were the main anthropic activity responsible for habitat loss and vegetation secondarization of *restinga* vegetation with 5622 ha. Since farming is intermittent, secondary forests occurred in similar proportions as anthropic activities (about 9000 ha each). Despite not providing quality habitat for specialist organisms, heterogeneous matrix areas occupied by secondary vegetation serve as corridors between fragments and provide important resources for more generalist species (Kuroe, Yamaguchi, Kadoya, & Miyashita, 2011). In this way, conservation and connectivity of habitats in the *Lençóis Maranhenses* can be achieved with adequate secondary vegetation management, such as adoption of sedentary farming practices that are less aggressive to biodiversity (Bhagwat, Kushalappa, Williams, & Brown, 2005).

Correlations between cover classes clearly indicate which kinds of habitats are more prone to be converted to agriculture: the dense *restinga* forests. The mapped territory occupied by secondary vegetation and anthropic activities was the result of deforestation of dense *restinga* forests. Considering that there are only 23,825.48 ha of dense *restinga* formations (CA-DENSE), and that CA-SECOND and CA-ANTRO together comprise 17,809.54 ha, we can say that at least 42% of the original vegetation cover has already been converted to anthropic use. Due to the environmental heterogeneity of the studied area and the inherent difficulties of mapping, which can lead to erroneous classifications (Lausch et al., 2015; Li & Wu, 2004), this percentage could be even greater. In fact, it is difficult to distinguish between the secondary vegetation and dense shrub *restinga* through satellite images, thus the CA-DENSE values could have been overestimated to the detriment of CA-SECOND.

Trails present in the studied area occuped 1142 ha, corresponding to 13% of CA-ANTRO. Small vegetation openings, such as unpaved roads, even with little traffic and canopy cover of adjacent fragments can hamper organism movement, especially forest interior specialists, thus affecting connectivity and gene flow between isolated populations (Laurance, Stouffer, & Laurance, 2004; Powell, Wolfe, Johnson, & Stouffer, 2016). The lack of control and planning for trail opening has already been cited as a threat to the herpetofauna of LMNP (Miranda, Lopes Costa, & Rocha, 2012). Hence, adequate trail planning and management are necessary to improve connectivity and biodiversity conservation in the region.

Villages totaled 745 ha in the study area and were more important than urban zones, which reached 242 ha in the mapped area, and corresponded to 8.72% of CA-ANTRO. This result is related to the absence of land regularization actions developed in the LMNP, which allowed the human population to increase in the protected area over the last three decades (Abakerli, 2001). Rocha, Drummond, and Ganem (2010)) cite land regularization as the main blockage for adequate biodiversity conservation in Brazilian national parks, such as the LMNP.

Correlation between CA-ANTRO, CA-DENSE, CA-SECOND, and SHDI indicates the increase in landscape texture caused by fragmentation of *restinga* habitat patches and the maintenance of a heterogeneous matrix (Griffith et al., 2000; Jurasinski & Beierkuhnlein, 2006). SHDI metric is sensitive to landscape structure and can vary depending on the region and the treatments (Castillo, García-Martin, Longares Aladrén, & de Luis, 2015). Satir and Erdogan (2016) observed that the conversion of land use to agriculture led to a decrease in SHDI value due to the substitution of native heterogeneous vegetation for a homogeneous matrix dominated by agriculture in Turkey. A similar result was obtained in Alta Garrotxa, Spain, where forest management yielded a reduction of SHDI in 50 years, indicating landscape homogenization (Vila Subirós, Ribas Palom, Varga Linde, & Llausàs Pascual, 2009). On the other hand, Heredia-Laclaustra, Frutor-Mejías, and Gonzalez-Hidalgo (2013)) observed that the abandonment of agricultural areas and reforestation increased landscape heterogeneity in the Pirineus region, Spain.

Although there is a positive correlation between SHDI and *in loco* biodiversity indices (Concepción et al., 2008), our results are cautious. Increased heterogeneity in the study area occurred through the destruction of pristine habitats with more resources for the persistence of sensitive organisms to human activities. Thus, it is more likely that SHDI increase results in a greater generalist species richness to the detriment of the specialist ones. Recent studies contend that fragmentation can be positive to increase biodiversity, as long as the same amount of habitat is maintained in the landscape (Fahrig, 2017; Melo, Sponchiado, Cáceres, & Fahrig, 2017; Seibold et al., 2017), which was not seen in our study.

Our local scale analysis demonstrated that 22.3% of AUs have < 40% of native vegetation cover, therefore near the fragmentation threshold that foresees significant biodiversity losses in landscapes with less than 30% of vegetation cover (Fahrig, 2001; Martensen et al., 2012). When the AU is below the fragmentation limit, the persistence of species populations that require high quality habitats is expected to be low. Barreto et al. (2010) observed that 43% of the Balsas river basin in Maranhão state presents < 30% of vegetation cover, thus possibly compromising birds and mammals conservation in the area (Barreto et al., 2012). On the other hand, our analysis showed that 26.4% of *AU*s have > 60% of cover. Thus we can say that the studied area is moderately preserved considering that critical and well-conserved *AU*s occur in similar proportions. However, the need for restoration of native vegetation in the most critical *AU*s is evident, in order to increase fragment size and reduce fragmentation effects in the most degraded *AU*s (Lindenmayer, Franklin, & Fischer, 2006).

The conversion of the land use in LMNP drastically reduced important habitats for biodiversity conservation, promoting the landscape phytophysiological impoverishment. Quality habitats such as *restinga* forests and riparian forests are being converted for human use and are suffering from secondarization. In fact, less structurally-complex habitats, such as scattered *restinga* shrub formations tend to increase its relative abundance in the landscape for not suffering use conversion. Populations of *Leopardus tigrinus* and *Lontra longicaudis*, which are threatened species that occur in the area (IUCN, 2016), are affected by habitat loss in the dense *restinga* because they are sensitive to human presence and dependent on large areas of preserved forest (Galant, Vasseur Drumond, Tremblay, & Bérubé, 2009; Lyra-Jorge, Ribeiro, Ciocheti, Tambosi, & Pivello, 2010; Nagy-Reis, Nichols, Chiarello, Ribeiro, & Setz, 2017; Triglia, Gómez, Cassini, & Túnez, 2016). Therefore, conservation actions aiming at improving landscape structure in the LNMP should focus on increasing *restinga* forest habitats in the landscape, maintenance of gallery forests, and gradual

recolonization of human populations outside the LMNP area.

4.2. Conservation strategies

Considering the inherently elevated costs of forest restoration, especially in extensive areas, it is important to plan adequate strategies to reach the conservation goals efficiently (Vieira, Holl, & Peneireiro, 2009). Bhagwat et al. (2005) highlight the role of informal reserves and agroforests for biodiversity conservation in the Western Ghates of India and recommend this strategy for conservation in regions where formal reserves are surrounded by a matrix of crop areas. However, to guarantee the conservation status intended for an Integral Protection Conservation Unit, agroforests must be implemented as a transitory measure to improve matrix quality at the short- and mid-term, and restore the native vegetation in the long-term (Vieira et al., 2009).

In AUs where native vegetation cover is more preserved (NV − COV > 60%), significant conservation actions for landscape management are not needed. Vegetation cover maintenance and reduction of human pressure on natural resources, thus enabling natural regeneration of the native vegetation, are enough. In these AUs, land regularization actions are more important than landscape management. In AUs with NV − COV < 60%, management becomes increasingly more important than land regularization. Commitment Terms (CT) with the community should be established aiming to implement Agroforestry Systems to promote agro-successional restoration and guarantee the protection of native vegetation fragments. CTs must focus on improving matrix quality, increasing vegetation cover, restoring riparian forests, and reducing soil conversion use (Lindenmayer et al., 2006). It is expected that vegetation cover restoration will increase the chances of biodiversity persistence in the landscape as a whole.

The conservation strategies presented have the potential to guide actions of LMNP managers, aiming at obtaining faster and more efficient results by concentrating efforts in specific AUs (Barreto et al., 2010). By involving the community in the process of restoration of the LMNP, instead of opposing to it, a window of opportunity is opened in the search for conservation of *Lençois Maranhenses*, at the same time as it assures the quality of life of the people inserted within this landscape. The implementation of Agroforestry Systems, in substitution of the conventional cassava (*Manihot esculenta*) intermittent farmsteads and sedentary cashew (*Anacardium orcidentale*) plantations, can help the development of human populations, and at the same time it improves the environmental quality of the matrix and hampers deforestation of conserved fragments (Lindenmayer et al., 2006; Nair, 2011).

Studies *in situ*, aiming at evaluating biodiversity *in loco* and investigating the traditional agricultural effects on it, are highly encouraged to subsidize our results, since moderate impacts that do not alter the landscape, such as overfishing, exploration of forestry goods and cattle raising, are difficult to be evaluated through landscape analysis. The implementation of a biodiversity monitoring program together with conservation strategies becomes fundamental to evaluate the effectiveness of the proposed actions.

5. Conclusion

The conversion of land use affects LNMP landscape structure by promoting increased fragmentation, *restinga* forest secondarization, and reduction of dense *restinga* habitats especially south of the studied area. Less diverse and complex habitats (open fields and scattered *restingas*) become proportionally more abundant in the landscape due to the use conversion of dense *restinga* areas, affecting the conservation of threatened species. Intense management within the protected area leads to restrictions to degrading anthropic activities and the incentive to carry out agricultural activities less harmful to biodiversity. The methods adapted from Barreto et al. (2010) to define conservation strategies were demonstrated to be adequate for LMNP management, to promote an increase in *AU*s native vegetation cover and thus improve biodiversity conservation in the region.

Appendix A

Table A1
Pearson correlation results between metrics. Dense restinga cover, CA-DENSE, scattered restinga, CA-SCATTER, mangroves, CA-MANG, restinga fields, CA-GRASS, secondary vegetation, CA-SECOND, human activities, CA-ANTRO, and Shannon Diversity Index, SHDI. Strong correlations (r > 0.65) are bolded. All correlations were statistically significant at the 5% level, with the exception of the one marked with *.

Metrics	SHDI	CA-ANTRO	CA-SECOND	CA-GRASS	CA-MANG	CA-SCATTER
CA-DENSE	0.65	0.66	0.50	0.10	0.00	0.30
CA-SCATTER	0.50	0.32	0.10	0.20	0.00	–
CA-MANG	0.20	−0.04*	−0.10	−0.10	–	–
CA-GRASS	0.4	0.14	−0.10	–	–	–
CA-SECOND	0.50	0.67	–	–	–	–
CA-ANTRO	0.66	–	–	–	–	–

Table A2
NV–COV frequency distribution in the AUs.

NV-COV (%)	Frequency	%
80-100	33	12.64
60-80	36	13.79
40-60	25	9.58
20-40	26	9.96
0-20	33	12.64
0	108	41.38
Total	261	100

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