



## The role of protected and unprotected forest remnants for mammal conservation in a megadiverse Neotropical hotspot

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## ARTICLE INFO

### Keywords:

Assemblage composition  
Biomass  
Patch size  
Relative abundance  
Species-area relationship  
Threatened species

## ABSTRACT

The Brazilian Atlantic Forest of Southern Bahia is a megadiverse region given its remarkable number of species and endemism. Despite being a priority region for biodiversity conservation, the role of protected and unprotected forest remnants for long-term species conservation is unknown. Here, we unveil the main patterns of occurrence and distribution of medium- and large-sized mammals in remnants of the Atlantic Forest of Southern Bahia, to generate subsidies for applied conservation strategies. We recorded mammals using camera-traps, active search, and/or line-transect surveys and complemented our species list with literature data. We thus obtained information on richness attributes, relative abundance, and biomass of mammal species per forest remnant, compared assemblages in protected and unprotected areas, and finally investigated both species-area and biomass-area relationships. From 72 forest remnants assessed, we recorded 45 mammal species, including 19 threatened locally. Protected areas were richer in species, especially concerning threatened ones, and concentrated most of the mammal biomass, which presented consistently low values for most areas. The positive and significant species-area and biomass-area relationships further corroborate these patterns since protected areas are larger in size. Despite the historic anthropogenic pressures, we conclude that Southern Bahia still harbors an expressive mammal diversity, with protected areas being critical to maintain most of the species' richness and biomass across the entire region. Nevertheless, small unprotected remnants (<100 ha) safeguard mammal species, including threatened ones, stressing their importance to maintain mammal assemblages in one of the most important hotspots of the entire biome.

## 1. Introduction

Protected areas are one of the most important mechanisms to slow down biodiversity loss (Godet and Devictor, 2018; Gray et al., 2016). Across the world, several ecosystems retain a great portion of their original biodiversity within these areas, preserving not only species, but also maintaining key ecological functions and ecosystem services performed by them (Bogoni et al., 2020a; Magioli et al., 2021). Nonetheless, protected areas alone cannot halt biodiversity loss (Godet and Devictor, 2018), given that at least a third of the world protected areas are current under intense human pressure (Jones et al., 2018). Moreover, the network of protected areas is poorly connected across most terrestrial ecosystems, while intact habitats are rapidly degrading due to emerging human pressures from surrounding areas, including deforestation and other anthropogenic disturbances (Ward et al., 2020). In fact, ecosystems integrity substantially decayed over the centuries, which resulted in less than 40% of forest ecosystems worldwide presenting high levels of ecosystem integrity, most of them located within strictly protected areas (Grantham et al., 2020).

Despite the importance of large protected areas, the conservation of small and unprotected habitat patches in private lands is an effective tool for improving the structural and functional connectivity of a landscape with a poorly-connected network of protected areas (Noss et al., 2012). There is a growing body of literature highlighting the importance of including unprotected habitat patches in private lands in conservation initiatives parallel with protected area networks (Capano et al., 2019; Gooden and 't Sas-Rolfes, 2020). Human-modified landscapes are dominant in most ecosystems worldwide, and generally composed of small habitat patches immersed in anthropogenic matrices, especially agricultural land uses in tropical forest regions (Gibbs et al., 2010). Although being characteristically less biodiverse than pristine areas, small forest remnants in these landscapes still retain a subset of the original biodiversity, play an important role in slowing down species loss and extinctions (Wintle et al., 2019), and are key areas when aiming

at conservation efforts in the long-term (Kremen and Merenlender, 2018). Most of these small and unprotected habitat patches are within private lands, which are required to be preserved if decision-makers intend to ensure species persistence in highly modified ecosystems (Lindenmayer, 2019), such as the Neotropical forests.

The Neotropical realm holds impressive levels of endemism and safeguards several world's terrestrial biodiversity hotspots, even though there are still many knowledge gaps that need to be fulfilled (Oliveira et al., 2016). The Brazilian Atlantic Forest is one of these extraordinary hotspots, given its remarkable number of species and endemism (Joly et al., 2014) combined to high levels of forest loss and degradation. Thus, this biome is currently listed among the most threatened ecosystems worldwide (Rezende et al., 2018). Several regions of the Atlantic Forest present scarce or low-quality information on species occurrence and distribution, including various protected areas (from strictly protected to sustainable use reserves), some of which, to date, are deprived of any basic information on their biodiversity. Primary data on species occurrence and distribution are required to subsidize studies on applied ecology, thereby supporting evidence-based conservation initiatives, especially because information on both protected and unprotected areas are still scant across many regions (Oliveira et al., 2017). Moreover, the amount and quality of this kind of information are also needed for a more assertive application of financial resources aiming to develop strategies and actions toward biodiversity conservation from local to biome-scale.

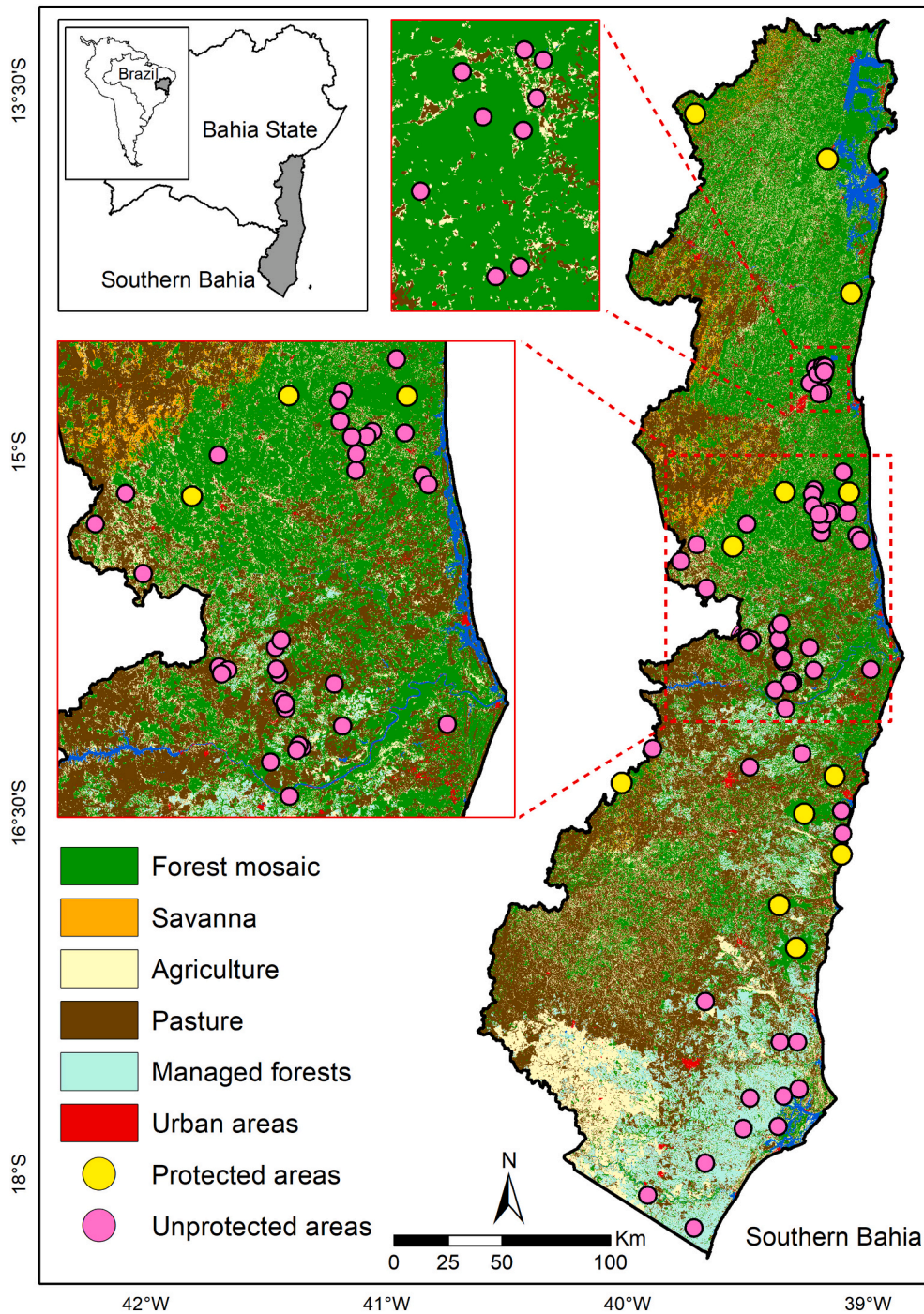
The Northeast portion of the Atlantic Forest is considered megadiverse, yet has been severely threatened by human activities (e.g., agriculture, mining, and logging), with only 1–2% of its original forest cover remaining (Cl et al., 2001; Costa and Guerra, 2012). Additionally, the Northeast region is underrepresented in the number of protected areas compared to the South and Southeast regions of this biome. The existing protected areas are small and isolated, compromising the maintenance of wildlife populations and ecological processes (Cl et al., 2001), and likely disrupting the provision of ecosystem services (Bogoni

et al., 2020a). Particularly, the Southern mesoregion coast of Bahia State (henceforth Southern Bahia) harbors at least 80% of all Atlantic Forest remnants of Northeast of Brazil (Ayres et al., 2005), and safeguards large and strictly protected areas, being considered top priority for conservation (MMA, 2018).

Among faunal species, mammals are considered a keystone group to boost conservation of entire landscapes worldwide. Indeed, mammals are considered umbrella species for conservation (Jenkins et al., 2013), and may encompass several taxonomic groups within their large home ranges when target of conservation strategies. Their importance for forest functioning is enormous, including herbivory, seed dispersal and

depredation, and vertebrate and invertebrate predation (Magioli et al., 2021), in addition to influencing carbon cycle through trophic interactions (Sobral et al., 2017). Yet several mammal species have been globally, regionally or locally extirpated in tropical biomes, such as the Atlantic Forest. Understanding their main threats is fundamental for scientists and policy-makers to propose sound conservation strategies to guarantee mammal persistence in the long-term and consequently the provision of key ecological functions.

Here, we unveil the main patterns of occurrence and distribution of medium- and large-sized mammals in forest remnants of the mega-diverse Atlantic Forest hotspot of Southern Bahia, Brazil. In particular,



**Fig. 1.** Study region in the Atlantic Forest of Southern Bahia, Brazil, showing the 72 forest remnants where assemblages of medium- and large-sized mammals were sampled. The main land use cover (Projeto MapBiomas, 2020) and the protected status of the forest remnants are also provided (see description of each site in Table B1 in Appendix B).

we (i) present a collective effort of governmental, non-governmental and private initiatives on the collection of primary data, combined with records compiled from the literature, to provide a comprehensive list of mammal species still occurring in forest remnants; (ii) compared mammal assemblages in terms of richness of overall species and threatened species, relative abundance and biomass between protected and unprotected areas; (iii) assessed if richness attributes comply with patch size thresholds identified for mammal assemblages in the Atlantic Forest; and (iv) perform both species-area and biomass-area relationships to evaluate if forest remnant size explains patterns of richness of overall and threatened species and biomass in this key region for mammal species conservation.

It has been hypothesized that species loss is more intense in smaller and more isolated habitat patches than expected by habitat loss alone due to changes in ecological processes, known as the 'ecosystem decay hypothesis' (Hanski et al., 2013; Lovejoy et al., 1984). Nonetheless, in landscapes where habitat loss took place more than one century ago, species turnover can compensate for the predicted species loss by introducing species previously absent in pristine areas (Chase et al., 2020). After five hundred years of colonization, the forests of Southern Bahia are highly degraded and modified, but the few forest remnants are still significant reservoirs of the regional biota (Faria et al., 2021). We expect protected areas in Southern Bahia, which are also the remaining largest forest patches, to retain the most complete mammal assemblages, including species more vulnerable to habitat loss and/or large-sized ones with higher overall relative abundances and biomass. On the other side, unprotected small forest remnants are expected to show opposite patterns. In addition, positive species- and biomass-area relationships are expected to be observed considering all studied forest remnants.

## 2. Material and methods

### 2.1. Study areas

The Southern Bahia is located within the Bahia biogeographical sub-region of the Atlantic Forest in Brazil, of which 92.3% of all forest fragments are smaller than 100 ha, indicating high levels of fragmentation and isolation (Ribeiro et al., 2009). Nonetheless, Southern Bahia remains a relatively well-preserved portion of this biome, where over 50% of the land is covered by forested mosaics, mainly shade cacao plantations (26% of the landscape), and secondary (19%) and primary forests (9%) (Landau et al., 2008). There is a wide variety of natural environments with different phytophysiognomies such as *florestas montanas*, *florestas de tabuleiro*, *restingas* and *muçumungas* (Pinto, 2015), but dense ombrophilous forests predominate (Peixoto et al., 2008). The main economic activities of this region are cattle ranching, agriculture (cocoa and coffee plantations), managed forests (particularly *Eucalyptus* spp. for pulp and paper), and tourism (Landau et al., 2008; MMA et al., 2006). The regional climate, according to Köppen classification, is hot and humid without a dry season. Mean annual temperature is 24 °C, and annual rainfall averages 2000 mm/year (Thomas et al., 1998).

The data we collected on medium- and large-sized mammals originated from 11 research projects and private environmental assessments, involving universities, governmental research centers, non-governmental organizations, and private companies. We conducted surveys between 2002 and 2020 in 68 forest remnants located at Southern Bahia (Fig. 1), including protected ( $N = 8$ ) and unprotected areas ( $N = 60$ ) within distinct landscape contexts. Forest remnant size ranged from 2 to 24,300 ha; description of the study remnants is detailed in Appendix A (Figs. A1-A10). To complement the species list obtained through primary data collection, we also searched the published literature for previous studies conducted in the region, considering indexed and non-indexed scientific articles, book chapters, and gray literature (e.g., scientific reports, and management plans of the protected areas) (Bahia, 1997, 2005; Cassano and Kierulff, 2009; Flesher, 2014; ICMBio, 2014, 2016; Moura, 2003; Ruschi, 1978; Sánchez-Lalinde et al., 2019).

We added another four protected areas from literature, ending up with a total of 72 forest remnants, being 12 protected and 60 unprotected areas. Detailed information about sampling areas can be seen in Appendix A, and Table B1 in Appendix B.

### 2.2. Sampling methods

Essentially, we employed three methods to survey mammal species: camera trapping, line transect surveys, and active search. Each study used different sampling designs, using either a single or combined method, resulting in distinct sampling efforts (see Appendix A). Camera trapping sampling design varied among sites, from 1 to 46 camera trap stations per site, with and without bait, and in some cases adopting specific sampling protocols [e.g., TEAM protocol (TEAM, 2011)]. Camera traps were deployed inside forest remnants, along trails, dirt roads and in the adjacent anthropogenic matrices; they were all installed at 30–40 cm from the ground, and operated 24 h per day. Sampling effort varied from 40 to 9263 trap-days.

For the line-transect method (Buckland et al., 1993; Peres and Cunha, 2011), the observer(s) walk on a linear transect (randomly placed or predetermined in a sampling area) at constant and low speed (~1 km/h). Whenever the observer detects an animal, the type of record (sighting or vocalization), time, distance from its current position to the animal, the angle of the detection to the transect line, number of individuals and species identification are recorded, in addition to the geographic location. We conducted samplings from dawn to dusk (distributed along the day), with transect lengths ranging from 100 to 7500 m. Active search method consists of walking on dirt roads and trails inside the forest remnants, at low speed (~1 km/h; Voss and Emmons, 1996), searching for direct (i.e., sightings, vocalizations, and carcasses) and indirect signs (i.e., tracks, feces, burrows, and food leftovers) of mammalian occurrence.

To identify the medium- and large-sized mammal species, we consulted specialized literature (Becker and Dalponte, 2013; Borges and Tomás, 2008; Emmons and Feer, 1997; Oliveira and Cassaro, 2006). We considered medium-sized mammals those weighing from 1 to 7 kg (Chiarello, 2000), whereas species weighing over 7 kg were considered large-sized (Emmons and Feer, 1997). We also included some species weighing less than 1 kg in our list because they are commonly recorded in medium- and large-sized mammal inventories (e.g., *Guerlinguetus brasiliensis*, *Callithrix* spp., *Leontopithecus chrysomelas*). Species nomenclature followed Abreu-Jr et al. (2020). We further assigned threat categories per species (i.e., Vulnerable, Endangered and Critically Endangered) following Bahia (2017), ICMBio (2018) and IUCN (2020) for regional, national and international levels, respectively. We considered exotic species those that occurred outside its natural range, as defined in Falk-Petersen et al. (2006).

### 2.3. Data analysis

First, we calculated the percentage of protected ( $N = 12$ ) and unprotected areas ( $N = 60$ ) in which each species occurred, and compared the difference in species richness between them, considering both overall species and threatened species, using the Wilcoxon rank-sum test. Then, considering all the assemblages in the dataset ( $N = 72$ ), we tested the relationship between the richness of both overall species and threatened species with the forest remnant size (henceforth patch size, in ha) using a generalized linear model (GLM, Poisson distribution). Variables were tested for normality using Shapiro-Wilk normality test, and patch size was log-transformed. For protected areas, we extracted the patch size from official regulatory documents, while for unprotected ones, we calculated their area using Google Earth Pro (patch size of each area is in Table B1 in Appendix B). Then, we tested spatial autocorrelation of the residuals from the GLMs using Moran I test from *ape* R package (Paradis and Schliep, 2019). We observed a positive spatial autocorrelation for both richness attributes (overall species and

threatened species), but only significant for the latter ( $p = 0.003$ ; overall species,  $p = 0.21$ ).

Given the spatial autocorrelation, we created buffers of 4.5 km radius from the center of each forest remnant, and grouped assemblages that presented overlap among buffer zones (Fig. C1 in Appendix C). We adopted this buffer size using the average home range of Neotropical medium- and large-sized mammals (Wilman et al., 2014) as reference, following the approach used by Bogoni et al. (2020b). We acknowledge that different species may respond to distinct scales of effect (Amiot et al., 2021), and that overlapping buffer zones may decrease, increase or not affect at all the spatial autocorrelation among assemblages (Zuckerberg et al., 2012), and therefore, should be carefully considered. In this study, as we were interested in the response of the assemblages, not focusing on single species, we adopted a single buffer size due to the variation in species richness (overall and threatened) and composition among the studied assemblages, which include taxa with distinct home range sizes, from small (e.g., sloths (*Bradypus* sp.)) to large (e.g., puma (*Puma concolor*)).

Finally, we tested the relationships between richness of both overall species and threatened species with patch size using generalized linear mixed models (GLMM, Poisson distribution), implemented through the *lme* R package (Bates et al., 2015), including the forest remnants as a random effect. The explanatory power of the GLMMs was measured by the marginal  $R^2$  ( $R^2_m$ ), which represents the variance explained by the fixed effects, the conditional  $R^2$  ( $R^2_c$ ), which is interpreted as the variance explained by both fixed and random effects, and  $p$  values using the *MuMIn* R package (Barton, 2020).

Since studies used different methods and presented distinct sampling efforts, we conducted the same analytical procedure previously described (comparison of richness attributes between protected and unprotected areas, relationship between richness attributes and patch size – GLMs and GLMMs, and spatial autocorrelation tests) using only camera trap data, given this sampling technique was the most used across forest remnants ( $N = 57$ ). For the GLM and GLMM models, we included the sampling effort as weight in the models, and tested variables for normality. For the GLMM, we also included the forest remnants as a random effect (Fig. C1 in Appendix C). We used these models as robustness tests for the relationships between richness of both overall species and threatened species with patch size considering all 72 mammal assemblages. Again, there was a positive spatial autocorrelation for both richness attributes (overall species and threatened species), but only significant for richness of threatened species ( $p = 0.02$ ; overall species,  $p = 0.37$ ).

From data collected through camera trapping ( $N = 57$ ), we used the capture rate of each species weighted by the sampling effort (number of individual records \* 100 / sampling effort) as a proxy of the relative abundance of species. We only considered records from the same species independent if taken at intervals longer than one-hour by the same camera trap. We recognize that the proxy used to estimate the relative abundance of mammals has limitations, since detection rate might not be directly related to abundance because species have different detection probabilities (Harmsen et al., 2010; Thompson et al., 1998; Tobler et al., 2008). However, this approach helps us to highlight common and rare species across sampling sites, and serve its purpose in assessing the variation in assemblage patterns between protected and unprotected areas.

We calculated the total biomass (in kg) of each assemblage by multiplying the relative abundance of each species by their mean adult body mass (based on Paglia et al., 2012), and then, summing all values generated for each mammal assemblage. For social species (e.g., pecararies and primates), we multiplied the biomass values by the mean number of individuals recorded in the camera traps and/or observed in the field, and compared to information on the literature (Emmons and Feer, 1997; Reis et al., 2006; species profiles in ICMBio Assessments of Extinction Risk - <https://www.icmbio.gov.br/portal/faunabrasileira/avaliacao-do-risco-de-extincao>). We compared assemblage biomass

between protected and unprotected areas using the Welch two-sample  $t$ -test. Following the analytical approach previously described, we performed a GLM to assess the relationship between assemblage biomass and patch size, considering the sampling effort as weight for the model; variables were log-transformed and tested for normality. Then, we tested the spatial autocorrelation of the residuals of the GLM, which presented a positive and significant spatial autocorrelation ( $p = 0.006$ ). Lastly, we performed a GLMM between biomass and patch size, including the forest remnants as a random effect (Fig. C1 in Appendix C) and using the sampling effort as weight for the model.

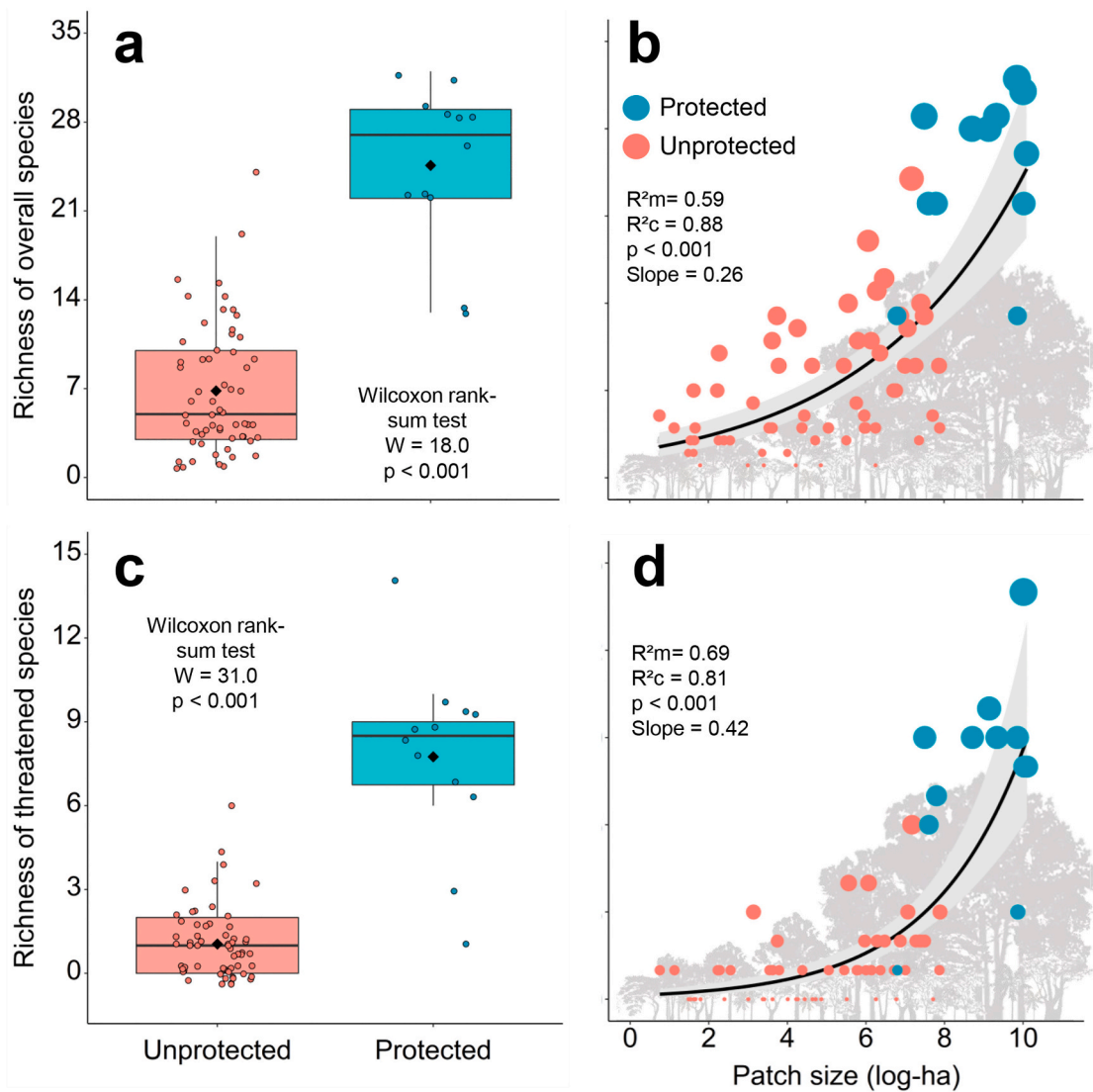
Finally, we compared the richness of both overall species and threatened species among the 72 mammal assemblages by grouping them according to the thresholds identified by Magioli et al. (2015) between functional diversity and patch size of medium- and large-sized mammals of the Atlantic Forest. The measure of functional diversity used by Magioli et al. (2015) is closely related to species richness, thus a good representation of thresholds between species richness and patch size. Therefore, we classified assemblages in three groups based on patch size thresholds – 1)  $< 60$  ha; 2) from 60 to 2050 ha; 3)  $> 2050$  ha – indicating assemblages with low richness, with richness increasing with increasing patch size, and assemblages in a more stable situation, respectively. We assessed differences among groups using Kruskal-Wallis rank-sum test and Dunn's test of multiple comparisons (dunn.test R package, Dinno, 2017). All analyses were performed in R 4.0.3 (R Core Team, 2020). We used the packages *ggplot2* (Wickham, 2016) for graphical implementation and *forcats* (Wickham, 2020) for data reordering.

### 3. Results

From the 72 forest remnants inventoried in Southern Bahia, we compiled a list of 45 native medium- and large-sized mammal species, belonging to 23 families and 8 orders (Table 1; Table B2 in Appendix B; Fig. C2 in Appendix C). Forty-two species were recorded from primary data, while three were only listed by records from the literature (*Callithrix penicillata*, *Myrmecophaga tridactyla* and *Priodontes maximus*). Considering only presence data for all assemblages, the most recorded species was the nine-banded armadillo (*Dasypus novemcinctus*; 67% of the forest remnants), followed by the South American coati (*Nasua nasua*; 58%) and the crab-eating fox (*Cerdocyon thous*; 54%) (Table 1).

The richness of overall species among forest remnants varied from 1 to 32 mammal species (mean  $9.8 \pm 8.4$  standard deviation), but was higher in protected areas ( $24.6 \pm 6.4$ ; Wilcoxon rank-sum test,  $W = 18.0$ ,  $p < 0.001$ ) than in unprotected ones ( $6.8 \pm 5.0$ ) (Fig. 2a). Considering all assemblages, richness of overall species presented a positive relationship with patch size ( $R^2_m = 0.59$ ,  $R^2_c = 0.88$ ,  $p < 0.001$ , slope = 0.26) (Fig. 2b). Even when considering only camera trap data ( $N = 57$ ), and including sampling effort as weight for the model, we still found differences in richness of overall species between protected and unprotected areas ( $W = 20.0$ ,  $p < 0.001$ ; Fig. C3a in Appendix C), and observed a positive relationship with patch size ( $R^2_m = 0.50$ ,  $R^2_c = 0.83$ ,  $p < 0.001$ , slope = 0.20; Fig. C3b in Appendix C). Additionally, species occurrence was proportionally higher in protected than unprotected areas, with those areas also showing larger sizes (Fig. 3).

In Bahia State, 42% of the native mammals listed in our study are threatened with extinction to some degree (Bahia, 2017); 38% of them in Brazil (ICMBio, 2018) and 29% worldwide (IUCN, 2020) (Table 1). Threatened species were more recorded in protected areas, occurring in 18 out of 72 sampling remnants. The margay (*Leopardus wiedii*) and the black-handed titi (*Callicebus melanochir*) were the threatened species most recorded (~25% of the forest remnants; Table 1). The richness of threatened species varied from 1 to 14 species, and was higher in protected areas ( $7.7 \pm 3.3$ ) than in unprotected ones ( $1.0 \pm 1.2$ ) (Fig. 2c). The richness of threatened species considering all assemblages presented a positive relationship with patch size ( $R^2_m = 0.69$ ,  $R^2_c = 0.81$ ,  $p < 0.001$ , slope = 0.42). Considering only camera trap data, the



**Fig. 2.** (a) Overall richness of medium- and large-sized mammal assemblages in protected ( $N = 12$ ) and unprotected areas ( $N = 60$ ) of the Atlantic Forest in Southern Bahia, Brazil. (b) Relationship between richness of overall species and patch size (log, in ha) of protected and unprotected areas ( $N = 72$ ). (c) Richness of threatened species in protected and unprotected areas and, (d) its relationship with patch size. Boxplots show means (diamonds), medians and quartiles and outliers. Dot size in (b) and (d) depicts richness of overall species and threatened species in each forest remnant, respectively; the shaded area represents the 95% confidence interval of the regressions.

differences on richness of threatened species between protected and unprotected areas were maintained ( $W = 25.5$ ,  $p < 0.001$ ; Fig. C3c in Appendix C), as well as its relationship with patch size ( $R^2m = 0.38$ ,  $R^2c = 0.80$ ,  $p < 0.001$ , slope = 0.34; Fig. C3d in Appendix C).

We recorded the presence of four domestic (*Bos taurus*, *Canis familiaris*, *Equus ferus caballus*, and *Felis catus*) and one exotic (*Lepus europaeus*) mammal species in forest remnants. Among domestic species, we recorded the domestic dog (*C. familiaris*) in ~47% of the sampling forest remnants (Table 1), which also comprised the fourth species most recorded in the region, occurring in a similar proportion on protected and unprotected areas (Fig. 3).

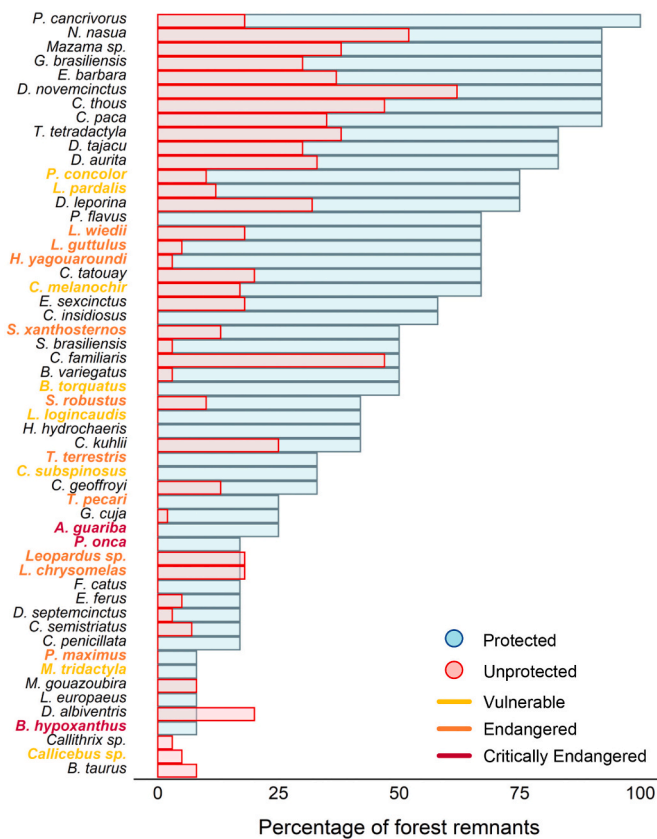
The relative abundance of each recorded species varied widely (0–4.28; Fig. 4); generalist species regarding habitat and diet presented the highest mean relative abundances (e.g., *D. novemcinctus*, *C. thous*, and *Didelphis aurita*), while most species presented low values and exhibited great variation among sampling sites. In protected areas, the relative abundance of species was higher for most mammals than in unprotected ones (Fig. C4 in Appendix C). The relative abundance of the domestic dog was high in both types of areas. Assemblage biomass also

varied among sampling sites, but was higher in protected areas ( $2486.5 \pm 1416.8$  kg; Welch two-sample  $t$ -test,  $t = -5.64$ ,  $df = 9.5$ ,  $p < 0.001$ ) than in unprotected ones ( $426.2 \pm 600.5$  kg) (Fig. 5a). Overall biomass presented a positive, but weak, relationship with patch size ( $R^2m = 0.07$ ,  $R^2c = 0.10$ ,  $p < 0.001$ , slope = 0.33) (Fig. 5b), with the Descobrimento National Park presenting the highest biomass value.

Comparing the richness of overall species and threatened species among the 72 assemblages grouped by patch size thresholds (< 60 ha; from 60 to 2050 ha; > 2050 ha), we observed significant differences among all three groups for both richness attributes (Kruskal-Wallis rank-sum test, overall richness,  $H_{(2)} = 24.86$ ,  $p < 0.001$ ; threatened species,  $H_{(2)} = 24.82$ ,  $p < 0.001$ ) (Fig. 6).

#### 4. Discussion

We showed that forest remnants are critical for maintaining mammal diversity in Southern Bahia, regardless of their protection status. We reveal that the network of protected areas has higher richness of overall species and threatened species and higher biomass than smaller patches.



**Fig. 3.** Percentage of protected (N = 12) and unprotected areas (N = 60) at the Atlantic Forest of Southern Bahia, Brazil, where each medium- and large-sized mammal species was recorded. Species listed in Bahia state red list (Bahia, 2017) are highlighted according to the threat category. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

The latter are often unprotected and located on private lands but, may serve as a baseline for reestablishing and/or improving the structural and functional connectivity of this poorly-connected network of protected areas, slowing biodiversity loss. Our results suggest that some species, including large-sized ones and those more vulnerable to habitat loss, are resilient enough to keep persisting throughout the centuries in human-modified landscapes, using them as effective or supplementary habitats (Magioli et al., 2019). Despite this optimistic scenario, most of the unprotected areas are severely defaunated and harbored less than 10 species (60% of the sampling sites), revealing a worrisome situation across the region, highlighting the importance of protected areas. Except for generalist species, large-sized species and those more vulnerable to habitat loss presented low relative abundance across the region, also indicating a collapse in overall mammal biomass. This pattern is especially evident for threatened mammals, which are virtually restricted to protected areas, a trend also observed in other regions of the Atlantic Forest (Bogoni et al., 2018; Magioli et al., 2015). A substantial portion of the species was detected in large forest remnants, particularly those under legal and strict protection, highlighting the critical role of protected areas in maintaining mammal diversity and their populations in the long-term. Nonetheless, some threatened species were recorded even in the smallest remnants, which need to be integrated in conservation initiatives.

#### 4.1. Overall assemblage patterns

The total species richness in our study represents 75% of all medium- and large-sized mammals expected to occur in the Atlantic Forest of

Bahia [N = 60; based on Graipel et al., 2017, Paglia et al., 2012 and the records in this study; Table B3 in Appendix B]. Following the patterns of mammals in human-modified landscapes from the Atlantic Forest (e.g., Bovo et al., 2018; Cassano et al., 2014; Lima et al., 2017; Magioli et al., 2016), species that are considered habitat and/or diet generalists (e.g., *D. novemcinctus*, *N. nasua* and *C. thous*) were more widespread and presented higher relative abundances than specialists or those more vulnerable to habitat loss (e.g., *Tapirus terrestris*, *Tayassu pecari*, and *Panthera onca*) in our studied sites. The variation in species relative abundances was clearer between protected and unprotected areas. Overall, protected areas are larger in size and less subject to anthropogenic pressures (e.g., poaching, logging), maintaining more diverse mammal assemblages and less variable trends in relative abundance. Conversely, mammal assemblages in small unprotected areas are impoverished and defaunated, and exhibit high variation of relative species abundances.

Overall, these findings concur with predictions of the ecosystem decay hypothesis, i.e., smaller habitats differed in species richness and abundances (which also compromise ecosystem functions) when compared to larger habitats, which can be a result of ecosystem decay (Chase et al., 2020). Our findings can also be a result of the flow of individuals from suitable habitat areas (e.g., large patches) to unfavorable areas (e.g., smaller patches), which is related to the mass effect concept (Leibold et al., 2004; Shmida and Wilson, 1985). Finally, despite covering a small fraction of the landscape, protected areas can be considered the remaining strongholds of source populations for most mammals in Southern Bahia, areas that may guarantee species long-term persistence (Dias, 1996). Small unprotected areas, particularly those immersed in agricultural landscapes, may offer short-term food resources or habitat for species, acting as attractive population sinks, apparently good for the individual, but compromising the population reproductive potential across time (Dias, 1996). Nonetheless, some species may persist in smaller habitat patches in response to the immigration of individuals from source populations nearby (Pulliam, 1988). Studies assessing further assemblage patterns and the dynamics of metacommunities are encouraged to produce better understanding of the state under which the observed assemblages persist.

The importance of large forest remnants is further revealed by the increase in overall mammalian richness, and threatened species richness, with increasing patch size, as observed in other studies in the Atlantic Forest (e.g., Banks-Leite et al., 2014; Magioli et al., 2015; Pardini et al., 2010). By grouping assemblages according to patch size thresholds (see Magioli et al., 2015), we observed that assemblage overall richness was very characteristic of what is expected by each threshold group, presenting clear differences in species number among them. For the richness of threatened species, groups below the second threshold (< 2050 ha) were similar in number of species, indicating that despite an increase of the overall richness as patch size increases, mammals that are more vulnerable to habitat loss and/or large-sized species were virtually absent from most of the small remnants. All threatened species and large-sized ones were restricted to large forest remnants, especially those under legal protection, highlighting the ubiquitous role of protected areas for maintaining mammal diversity, and the need to connect them to smaller remnants to promote species movement and genetic flow.

The biomass of most assemblages was characteristically low (< 500 kg; 68% of the forest remnants), particularly in unprotected areas (38 out of 51 remnants sampled with camera traps), which can be explained by the absence of large-sized mammal species and/or their occurrence in extremely low relative abundance. Following patterns of species richness and relative abundance, overall biomass was also higher in protected areas. In particular, one protected area (Descobrimento National Park) showed a great overall biomass, which can be explained by the presence of white-lipped peccaries (*Tayassu pecari*) – a species that represents most of the mammal biomass in Neotropical forests where it occurs (Galetti et al., 2017; Peres, 2000). Although biomass increased

Table 1

Medium- and large-sized mammals recorded in 72 remnants at the Atlantic Forest of Southern Bahia, Brazil, including species trophic guild (Paglia et al., 2012), threat category at regional (Bahia, 2017), national (ICMBio, 2018) and international levels (IUCN, 2020), and the number of sites where each species was recorded.

Taxon	Common name	Guild	Threaten category			Remnants (N = 72)
			Bahia	Brazil	World	
<b>DIDELPHIMORPHIA</b>						
<b>DIDELPHIDAE</b>						
<i>Didelphis albiventris</i> Lund, 1840	White-eared opossum	Fr/Om				13
<i>Didelphis aurita</i> (Wied-Neuwied, 1826)	Black-eared opossum	Fr/Om				30
<b>PILOSA</b>						
<b>BRADYPODIDAE</b>						
<i>Bradypus variegatus</i> Schinz, 1825	Brown-throated sloth	Hb				8
<i>Bradypus torquatus</i> Illiger, 1811	Maned three-toed sloth	Hb	VU	VU	VU	6
<b>MYRMECOPHAGIDAE</b>						
<i>Myrmecophaga tridactyla</i> Linnaeus, 1758 <sup>a,c</sup>	Giant anteater	In	VU	VU	VU	1
<i>Tamandua tetradactyla</i> (Linnaeus, 1758)	Southern tamandua	In				33
<b>CINGULATA</b>						
<b>CHLAMYPHORIDAE</b>						
<i>Cabassous tatouay</i> (Desmarest, 1804)	Greater naked-tailed armadillo	In				20
<i>Euphractus sexcinctus</i> (Linnaeus, 1758)	Six-banded armadillo	In/Om				18
<i>Priodontes maximus</i> (Kerr, 1792) <sup>a,c</sup>	Giant armadillo	In	EN	VU	VU	1
<b>DASYPODIDAE</b>						
<i>Dasyops novemcinctus</i> Linnaeus, 1758	Nine-banded armadillo	In/Om				48
<i>Dasyops septemcinctus</i> (Linnaeus, 1758)	Seven-banded armadillo	In/Om				4
<b>PERISSODACTYLA</b>						
<b>EQUIDAE</b>						
<i>Equus ferus caballus</i> Boddaert, 1785 <sup>d</sup>	Domestic horse	Hb				5
<b>TAPIRIDAE</b>						
<i>Tapirus terrestris</i> (Linnaeus, 1758)	Lowland tapir	Hb/Fr	EN	VU	VU	4
<b>ARTIODACTYLA</b>						
<b>BOVIDAE</b>						
<i>Bos taurus</i> Linnaeus, 1758 <sup>d</sup>	Cattle	Hb				5
<b>CERVIDAE</b>						
<i>Mazama gouazoubira</i> (Fischer, 1814)	Gray brocket deer	Fr/Hb				6
<i>Mazama</i> sp. (Erxleben, 1777)	Brocket deer	Hb/Fr				34
<b>TAYASSUIDAE</b>						
<i>Dicotyles tajacu</i> (Linnaeus, 1758)	Collared peccary	Fr/Hb				28
<i>Tayassu pecari</i> (Link, 1795)	White-lipped peccary	Fr/Hb	EN	VU	VU	3
<b>PRIMATES</b>						
<b>ATELIDAE</b>						
<i>Alouatta guariba guariba</i> (Humboldt, 1812)	Northern brown howler monkey	Hb/Fr	CR	CR	CR	3
<i>Brachyteles hypoxanthus</i> (Kuhl, 1820)	Northern muriqui	Fr/Hb	CR	CR	CR	1
<b>CALLITRICHIDAE</b>						
<i>Callithrix geoffroyi</i> (Humboldt, 1812)	Geoffroy's tufted-ear marmoset	Fr/In				12
<i>Callithrix kuhlii</i> Coimbra-Filho, 1985	Wied's marmoset	Fr/In				20
<i>Callithrix</i> sp. Erxleben, 1777	Marmoset	Fr/In				2
<i>Callithrix penicillata</i> (É. Geoffroy, 1812) <sup>a</sup>	Black-pencilled marmoset	Fr/In				2
<i>Leontopithecus chrysomelas</i> (Kuhl, 1820)	Golden-headed lion tamarin	Fr/In	EN	EN	EN	13
<b>CEBIDAE</b>						
<i>Sapajus robustus</i> (Kuhl, 1820)	Crested capuchin	Fr/Om	EN	EN	EN	11
<i>Sapajus xanthosternus</i> (Wied-Neuwied, 1826)	Buff-headed capuchin	Fr/Om	EN	EN	CR	14
<b>PITHECIIDAE</b>						
<i>Callicebus melanochir</i> (Wied-Neuwied, 1820)	Black-handed titi	Fr/Hb	VU	VU	VU	18
<i>Callicebus</i> sp. Thomas, 1903	Titi monkey	Fr/Hb				3
<b>CARNIVORA</b>						
<b>CANIDAE</b>						
<i>Canis lupus familiaris</i> (Linnaeus, 1758) <sup>d</sup>	Domestic dog	Ca/Om				34
<i>Cercocyon thous</i> (Linnaeus, 1766)	Crab-eating fox	Om				39
<b>FELIDAE</b>						
<i>Felis catus</i> (Linnaeus, 1758) <sup>d</sup>	Domestic cat	Ca				2
<i>Herpailurus yagouaroundi</i> (É. Geoffroy Saint-Hilaire, 1803)	Jaguarundi	Ca	VU	VU		11
<i>Leopardus guttulus</i> (Schreber, 1775)	Southern tiger cat	Ca	EN	VU	VU	16
<i>Leopardus pardalis</i> (Linnaeus, 1758)	Ocelot	Ca	VU			18
<i>Leopardus wiedii</i> (Schinz, 1821)	Margay	Ca	EN	VU		13
<i>Leopardus</i> sp. Gray, 1842	Small spotted cat	Ca				2
<i>Panthera onca</i> (Linnaeus, 1758)	Jaguar	Ca	CR	VU		15
<i>Puma concolor</i> (Linnaeus, 1771)	Puma	Ca	VU	VU		10
<b>MEPHITIDAE</b>						
<i>Conepatus semistriatus</i> (Boddaert, 1785)	Striped hog-nosed skunk	In/Om				6
<b>MUSTELIDAE</b>						
<i>Eira barbara</i> (Linnaeus, 1758)	Tayra	Fr/Om				33
<i>Galictis cuja</i> (Molina, 1782)	Lesser grison	Ca				4
<i>Lontra longicaudis</i> (Olfers, 1818)	Neotropical otter	Ca	VU			5

(continued on next page)



Table 1 (continued)

Taxon	Common name	Guild	Threaten category			Remnants (N = 72)
			Bahia	Brazil	World	
PROCYONIDAE						
<i>Nasua nasua</i> (Linnaeus, 1766)	South America coati	Fr/Om				42
<i>Potos flavus</i> (Schreber, 1774)	Kinkajou	Fr/Om				8
<i>Procyon cancrivorus</i> (G. Cuvier, 1798)	Crab-eating raccoon	Fr/Om				23
LAGOMORPHA						
LEPORIDAE						
<i>Lepus europaeus</i> Pallas, 1778 <sup>b</sup>	European hare	Hb				1
<i>Sylvilagus brasiliensis</i> (Linnaeus, 1758)	Brazilian rabbit	Hb				8
RODENTIA						
CAVIIDAE						
<i>Hydrochoerus hydrochaeris</i> (Linnaeus, 1766)	Capybara	Hb				5
CUNICULIDAE						
<i>Cuniculus paca</i> (Linnaeus, 1766)	Lowland paca	Fr/Hb				32
DASYPROCTIDAE						
<i>Dasyprocta leporina</i> (Linnaeus, 1758)	Red-rumped agouti	Fr				28
ERETHIZONTIDAE						
<i>Chaetomys subspinosus</i> (Olfers, 1818)	Thin-spined porcupine	Hb	VU	VU	VU	4
<i>Coendou insidiosus</i> (Lichtenstein, 1818)	Bahia hairy dwarf porcupine	Fr/Hb				7
SCIURIDAE						
<i>Guerlinguetus brasiliensis</i> (Thomas, 1901)	Brazilian squirrel	Fr				29
Total (only native species)	45 (3 <sup>a</sup> )	-	19	17	13	-

Fr = frugivore; Om = omnivore; In = insectivore; Hb = herbivore; Ca = carnivore; VU = vulnerable; EN = Endangered; CR = Critically Endangered.

<sup>a</sup> Species only in the literature.

<sup>b</sup> Exotic species.

<sup>c</sup> Only historical records.

<sup>d</sup> Domestic species.

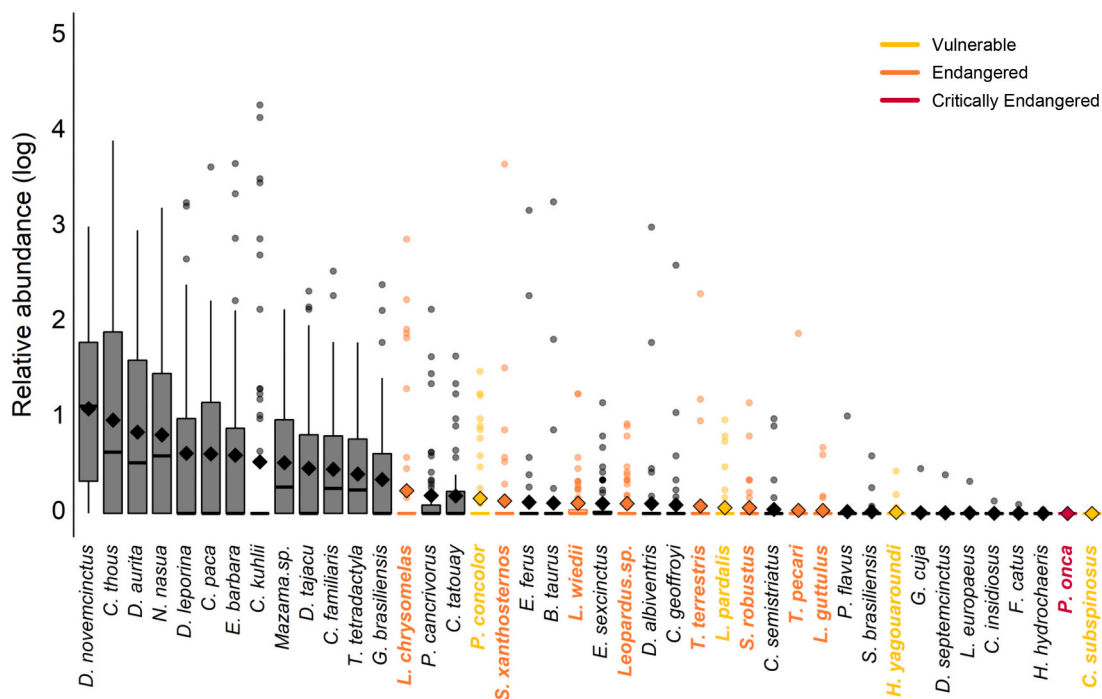
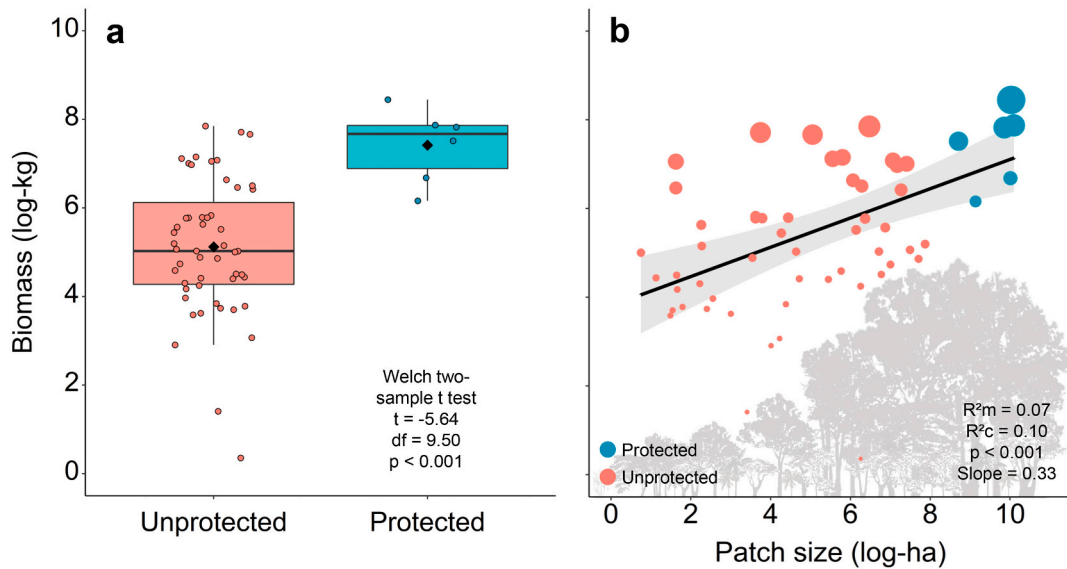
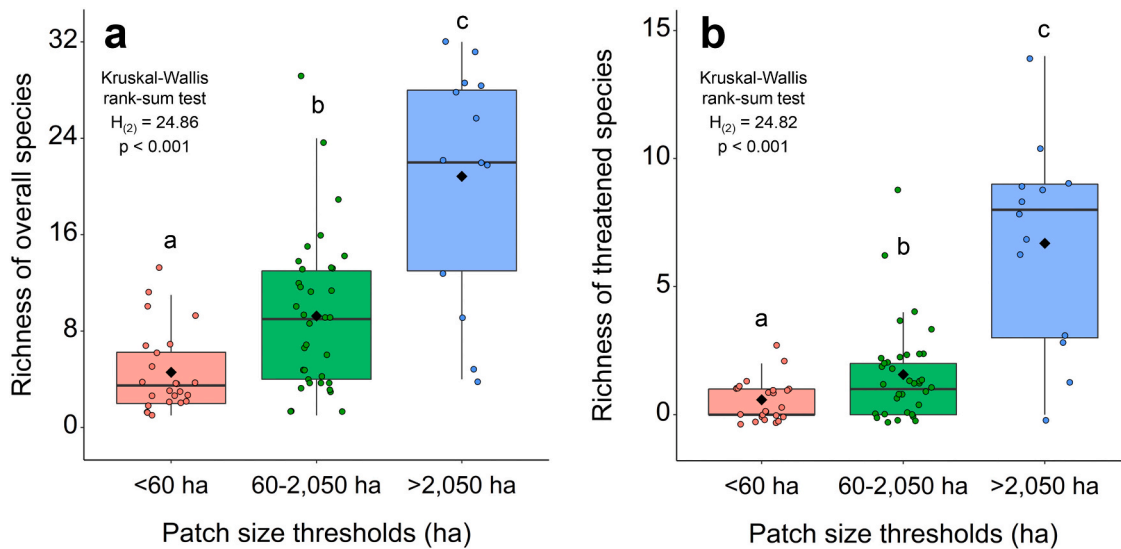


Fig. 4. Relative abundance (log-transformed) of medium- and large-sized mammals sampled with camera trapping in remnants (N = 57) of the Atlantic Forest of Southern Bahia, Brazil. Boxplots show means (diamonds), medians, quartiles and outliers. Species listed in Bahia state red list (Bahia, 2017) are highlighted according to the threat category. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)



**Fig. 5.** (a) Biomass (log, in kg) of medium- and large-sized mammal assemblages in protected ( $N = 6$ ) and unprotected areas ( $N = 51$ ) of the Atlantic Forest of Southern Bahia, Brazil, sampled with camera trapping. Boxplots show means (diamonds), medians, quartiles and outliers. (b) Relationship of assemblage biomass (log, in kg) and patch size (log, in ha) of protected and unprotected areas ( $N = 57$ ). Dot size depicts the biomass value of each assemblage, and the shaded area represents the 95% confidence interval of the regression.



**Fig. 6.** Comparison of richness of overall species and threatened species of medium- and large-sized mammal assemblages in the Atlantic Forest of Southern Bahia, Brazil. Assemblages were grouped by patch size thresholds (< 60 ha; from 60 to 2050 ha; > 2050 ha) as presented in Magioli et al. (2015). Lowercase letters represent significant differences among groups ( $p < 0.05$ ).

with patch size, the relationship was weak, with values similarly low for most areas. This trend has been observed throughout the Atlantic Forest (Canale et al., 2012; Galetti et al., 2009) and across the entire Neotropical realm (Bogoni et al., 2020b). Consequently, either local extinctions or reduced abundance of large-sized mammal species, i.e., defaunation process, can trigger a myriad of cascade effects in those forest remnants, ultimately affecting forest functionality. This includes a reduction, or even an interruption, in the prevalence of ecological functions (Magioli et al., 2021) and provision of ecosystem services (Bogoni et al., 2020a).

4.2. Species-specific patterns

From the species only mentioned in the literature, the black-

pencilled marmoset (*C. penicillata*) may occur in other forest remnants across Southern Bahia that were not sampled, similarly to other species with few records [e.g., capybara (*Hydrochoerus hydrochaeris*), sloths (genus *Bradypus*), other primates (genus *Callithrix*, *Callicebus* and *Sapajus*), kinkajou (*Potos flavus*), Neotropical otter (*Lontra longicaudis*) and thin-spined porcupine (*Chaetomys subspinosus*)]. The aforementioned species were possibly underrepresented because the main sampling method employed (i.e., camera trapping deployed on the ground) is not adequate to record them. For the larger insectivores, i.e., giant anteater (*Myrmecophaga tridactyla*) and giant armadillo (*Priodontes maximus*), records were rare. The giant armadillo is considered naturally rare in the Atlantic Forest (Srbek-Araujo et al., 2009), and is possibly regionally extinct in Southern Bahia (Chiarello et al., 2015). This species is

mentioned in interviews of a previous management plan of the RPPN Estação Veracel (Almeida et al., 1998), but there is no official record confirming its occurrence presently. Similarly, although the original distribution of the giant anteater overlaps a small portion of Southern Bahia (Miranda et al., 2015), historical records are mentioned in the literature only twice [Monte Pascoal National Park and Una Biological Reserve; (Ruschi, 1978; Santos et al., 2019)], with no recent records. Nevertheless, giant armadillo has previously been classified as area-sensitive species in non-poached forest remnants within a fragmented landscape in the Brazilian Amazonia, demonstrating its higher vulnerability to area reduction (Benchimol and Peres, 2015).

Other important large-sized species such as jaguars (*Panthera onca*), Northern muriquis (*Brachyteles hypoxanthus*), lowland tapirs (*Tapirus terrestris*) and white-lipped peccaries have virtually vanished from the region. Records of a single jaguar individual were obtained for the RPPN Estação Veracel (2016–2017) after more than 20 years without any official record of the species (Casanova et al., 2018); however, no further record was obtained in subsequent years (2018–2019). There are historical records of the Northern muriqui in the region (Culot et al., 2019), but its current presence was only confirmed in the Alto Cariri National Park and REVIS Mata dos Muriquis, two contiguous protected areas located in the interstate border between Minas Gerais and Bahia states (Melo et al., 2018). This primate species is virtually extinct in other parts of Bahia (Ferraz et al., 2019; Melo et al., 2004). The white-lipped peccary was recorded only in Descobrimento National Park, with historical records in other two areas (Monte Paschoal and Pau Brasil National Parks; ICMBio, 2016; Ruschi, 1978), but it is possibly extinct elsewhere in the region. Likewise, the same was observed for the lowland tapir, which was only recorded in three protected areas (Descobrimento and Pau Brasil National Parks, and RPPN Estação Veracel). The species has historical records in Monte Pascoal National Park (Ruschi, 1978), but has not been recently recorded. Considering that poaching off-takes comprise one of the strongest drivers of local extinctions in human-modified landscapes of Bahia (Canale et al., 2012), allied to the pervasive effect of habitat loss on mammal diversity (Dirzo et al., 2014), the disappearance of these large species can be potentially explained by the intense poaching in the region and centuries of habitat modification.

For the genus *Mazama*, previous studies mentioned the occurrence of both the red (*M. americana*) and gray brocket (*M. gouazoubira*) in the region. However, new evidence shows that the red brocket occurrence is limited to the Northern part of Amazonas river (Cifuentes-Rincón et al., 2020), thus, not occurring in the Bahia state. The Amazonian brocket (*M. nemorivaga*) was recently identified in Southern Bahia, specifically in Una Biological Reserve (Oliveira et al., 2020), a species supposed to occur only in Amazonia. Therefore, the Amazonian brocket is possibly one of the forest-dependent deer species that occurs in Southern Bahia instead of the red brocket. Additionally, some studies have mentioned the presence of the lesser naked-tailed armadillo (*Cabassous unicinctus*) in the region (e.g., Falcão et al., 2012; Moura, 2003), but as pointed out by Ribeiro et al. (2013), the greater naked-tailed armadillo (*C. tatouay*) is the species that possibly occurs therein, as also suggested by Anacleto et al. (2006).

Felids were poorly recorded across the region, exhibiting low relative abundances. In protected areas, pumas (*Puma concolor*) and margays (*L. wiedii*) have slightly more records, but elsewhere, felids frequently presented few records and low relative abundance. Particularly, ocelots (*Leopardus pardalis*) were recorded in few forest remnants with very low number of records per remnant (mostly, one or two), which is very contrasting with other parts of the Atlantic Forest where it is more commonly recorded (Lima et al., 2017). The striped hog-nosed skunk (*Conepatus semistriatus*) was recorded in four of our sampling sites, in addition to other records in the region (see Nagy-Reis et al., 2020). This species is uncommon in densely forested areas and seems to be expanding its distribution area toward the Atlantic Forest (Magioli et al., 2020).

#### 4.3. Anthropogenic current threats

The most evident threats to biodiversity in Southern Bahia are habitat loss, illegal logging, and poaching (Schiavetti et al., 2012). During data collection, we frequently observed direct and/or indirect evidence of the aforementioned threats throughout the region, in both protected and unprotected areas. Habitat loss is a main driver of species erosion in tropical regions, which drastically affect large-sized species, such as large herbivores (Ripple et al., 2015). This trend is evident in Southern Bahia, which presented substantial rates of deforestation in the last years (Fundação SOS Mata Atlântica and INPE, 2020), with large herbivores being virtually restricted to large remnants and those under legal protection.

In addition to the effects of habitat loss, poaching has been inducing substantial changes in the richness and abundance of mammals in the entire Atlantic Forest (Canale et al., 2012; Carvalho Jr. and Morato, 2013; Galetti et al., 2017). Indeed, we frequently observed poaching evidence during sampling, such as tree stands ('esperas'), baits (commonly composed by fleshy fruits), evidence of use of handmade cannons ('trabucos'), and camping sites. In response to elevated poverty levels in Northeast Brazil, poaching pressure is also high in the region (Canale et al., 2012; Castilho et al., 2017; Flesher and Laufer, 2013; Santos et al., 2018), which is influenced by the proportion of anthropogenic matrices surrounding the forest remnants, and the proximity to urban areas and human settlements (Santos et al., 2018).

Finally, the presence of non-native species also affects mammal native species in forest remnants. For instance, the presence of domestic dogs has been enhanced with the expansion and intensification of agricultural lands, and the increase of human settlements and urban areas, which were present in almost half of the sampling remnants. This species is responsible for a myriad of negative effects on biodiversity, such as predation, competition and disease transmission (Hughes and Macdonald, 2013; Young et al., 2011), potentially impacting mammals in both protected and unprotected areas of Southern Bahia.

#### 5. Implications for conservation

Our findings reinforce that protected areas in Southern Bahia are the last standing pillars for maintaining mammal diversity and their populations, showing that the mosaic of protected areas in this region fulfills its role in preserving biodiversity. Strictly protected areas safeguard most of the sensitive mammals and large-sized species in Southern Bahia, as noted by other studies (e.g., Ferreira et al., 2020). Nonetheless, protected areas of sustainable use also perform an important role in maintaining mammal diversity (see results for RPPN Estação Veracel and Michelin Ecological Reserve; Tables B1 and B2 in Appendix B). We have added knowledge on mammal occurrence on six protected areas, strengthening their importance as key areas for mammal conservation. Yet, some strictly protected areas (e.g., Alto Cariri, Boa Nova and Serra das Lontras National Parks) lack basic information on the existing biodiversity including mammals, which is also true for many protected areas of sustainable use and private reserves spread out across Southern Bahia. Further investments are necessary to maintain the integrity of protected areas, and the implementation of a framework designed to monitor and restrain illegal activities (e.g., poaching and logging), control fires and raise awareness of local communities of their importance. Therefore, supporting scientific research and integrating local people to increase knowledge on the extant biodiversity is desirable (Roque et al., 2018).

The network of protected areas across Southern Bahia is of great importance, but its structural connectivity is very limited, reflecting a worldwide trend for terrestrial ecosystems (Ward et al., 2020). Most of the region is occupied by private lands dominated by anthropogenic land uses, especially agriculture and pastures, which lead to complex cross-boundary interactions to promote connectivity between protected and unprotected areas (Blanco et al., 2020). It is essential to understand

the interface processes between local people and private landowners with protected areas to delineate interactive zones, aiming to understand the trade-offs (positive and negative effects) of their existence (Blanco et al., 2020). A reconciliation between conservation initiatives, production, local people and protected areas is undeniably necessary since mammals inhabit both natural and anthropogenic areas, using them as habitat and foraging sites (Magioli et al., 2019).

Despite presenting less diverse and impoverished mammal assemblages, small forest remnants still play a role of utmost importance (Fahrig, 2020), acting as stepping stones for the fauna to move between large remnants and those under legal protection, and preventing the local extinction of many species (Wintle et al., 2019). Some unprotected areas presented high species richness including many threatened ones, highlighting the need to also effectively preserve and protect these areas, especially those with potential to increase connectivity with larger remnants. Unprotected habitat patches within private lands are important features to be considered in conservation initiatives (Capano et al., 2019; Gooden and 't Sas-Rolfes, 2020) since protected areas are not enough to secure the persistence of threatened and endemic species, or those suffering population declines across their distribution (Clancy et al., 2020; Oliveira et al., 2017). In the Atlantic Forest, reestablishing structural connectivity may be achieved through vegetation restoration of Legal Reserves and Permanent Preservation Areas (e.g., riparian forests), which are legal instruments of the Brazilian Native Vegetation Protection Law of 2012 that are mandatory for most private lands. Pastures that are less-suitable for agriculture are good targets for restoration because they do not impair productive lands (Mello et al., 2021), and are a widespread land use in Southern Bahia (see Fig. 1).

Therefore, to support biodiversity conservation in this megadiverse region, we recommend future studies in both protected and unprotected areas across Southern Bahia, aiming to increase knowledge on mammalian occurrence and distribution. Some forest remnants were subsampled in our study, including both protected and unprotected areas, which reinforces the need for investments on basic research, aiming at longer sampling periods using complementary methods. In this sense, the implementation of long-term monitoring programs, as already occurring in other Brazilian protected areas (Costa-Pereira et al., 2013), are highly recommended to enhance our understanding on population fluctuations across time. Specifically, protected areas should be target of new studies (e.g., Alto Cariri, Boa Nova and Serra das Lontras National Parks), which normally are large forest remnants that lack long-term studies to properly address their importance for mammal conservation, and also considering that most protected areas in Brazil are subsampled, particularly those created more recently (Oliveira et al., 2017). Threatened species were recorded even in small unprotected remnants (from 2 to 100 ha; Table B2 in Appendix B), stressing that human-modified landscapes should also be target of new studies, in order to better direct restoration actions toward halting the loss of species, ecological functions and the provision of ecosystem services (Bogoni et al., 2020a; Chase et al., 2020; Magioli et al., 2021).

Supplementary data to this article can be found online at <https://doi.org/10.1016/j.biocon.2021.109173>.

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**Otávia Crepaldi:** Data collection, Funding acquisition, Writing - Reviewing and Editing; **Lúcia Ângelo Machado Mendes:** Data collection, Writing - Reviewing and Editing; **Rodrigo de Almeida Nobre:** Data collection, Funding acquisition, Writing - Reviewing and Editing; **Adriano Garcia Chiarello:** Data collection, Funding acquisition, Writing - Reviewing and Editing; **Alvaro García-Olaechea:** Data collection, Writing - Reviewing and Editing; **Andreza Bellotto Nobre:** Data collection, Writing - Reviewing and Editing; **Camila Cantagallo Devids:** Data collection, Writing - Reviewing and Editing; **Camila Righetto Cassano:** Data collection, Funding acquisition, Writing - Reviewing and Editing; **Christine Del Vecchio Koike:** Data collection, Writing - Reviewing and Editing; **Christine Steiner São Bernardo:** Data collection, Funding acquisition, Writing - Reviewing and Editing; **Daniel Henrique Homem:** Data collection, Writing - Reviewing and Editing; **Daniel da Silva Ferraz:** Data collection, Writing - Reviewing and Editing; **Diego Leal Abreu:** Data collection, Writing - Reviewing and Editing; **Eliana Cazetta:** Funding acquisition, Writing - Reviewing and Editing; **Elson Fernandes de Lima:** Funding acquisition, Writing - Reviewing and Editing; **Fernando César Gonçalves Bonfim:** Data collection, Writing - Reviewing and Editing; **Fernando Lima:** Data collection, Writing - Reviewing and Editing; **Helena Alves do Prado:** Data collection, Writing - Reviewing and Editing; **Henrique Gonçalves Santos:** Data collection, Writing - Reviewing and Editing; **Joana Zorzal Nodari:** Data collection, Writing - Reviewing and Editing; **João Gabriel Ribeiro Giovanelli:** Data collection, Funding acquisition, Writing - Reviewing and Editing; **Marcello Silva Nery:** Data collection, Writing - Reviewing and Editing; **Michel Barros Faria:** Data collection, Writing - Reviewing and Editing; **Priscila Coutinho Ribas Ferreira:** Data collection, Writing - Reviewing and Editing; **Priscilla Sales Gomes:** Funding acquisition, Writing - Reviewing and Editing; **Raisa Rodarte:** Data collection, Writing - Reviewing and Editing; **Rodrigo Borges:** Data collection, Writing - Reviewing and Editing; **Thais Fantini Sagrillo Zuccolotto:** Funding acquisition, Writing - Reviewing and Editing; **Tathiane Santi Sarcinelli:** Funding acquisition, Writing - Reviewing and Editing; **Whaldener Endo:** Data collection, Writing - Reviewing and Editing; **Yugo Matsuda:** Funding acquisition, Writing - Reviewing and Editing; **Virgínia Londe de Camargos:** Funding acquisition, Writing - Reviewing and Editing; **Ronaldo Gonçalves Morato:** Conceptualization, Funding acquisition, Writing - Reviewing and Editing.

#### Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

#### Acknowledgements

This paper is a result of Rede de Pesquisas de Conservação da Biodiversidade do Corredor Central da Mata Atlântica (REDE CONBIO). We thank the teams of Pau Brasil National Park and RPPN Estação Veracel, Edmilson Figueiredo, Edney Santos dos Reis, Adenildo Quinto Xavier, Robenilton Silva da Cruz, Mario Cirri e Lapique do Brasil Empreendimentos Imobiliários Ltda for technical support in the field; Instituto Chico Mendes de Conservação da Biodiversidade - ICMBio (research authorization n. 30926); Instituto do Meio Ambiente e Recursos Hídricos da Bahia - INEMA (research authorization n. 2013-006433/TEC/PESQ-0027).

#### Funding

This work was supported by Centre for Research and Conservation, Royal Zoological Society of Antwerp; Centro Nacional de Desenvolvimento Científico e Tecnológico [grant number 181905/2002-7]; Fundação Biodiversitas (Programa de Espécies Ameaçadas); Fundação Biodiversitas and CEPAN [grant number 028M/012004]; Fundação de

Amparo à Pesquisa do Estado da Bahia (FAPESB) [grant numbers 2366/2012, 1760/2013]; Fundação Grupo Boticário de Proteção à Natureza [grant number 0939-20121]; Idea Wild; Instituto Arapyaú; National Geographic Society [grant number EC-335R-18]; Programa de Conservação da Biodiversidade da Barra do Tijuípe - BA [project number 05]; Projeto de Conservação e Utilização Sustentável da Diversidade Biológica Brasileira (PROBIO) [grant number 68.0033/02-5]; Rufford Small Grants for Nature Conservation [grant numbers 11495-1, 19666-1]; SAVE Brasil; Universidade Estadual de Santa Cruz (PROPP) [grant number 413 00220.1100.1840]; Universidade Estadual do Sudoeste da Bahia (UESB - Setor de Transportes - Campus Jequié); Veracel Celulose SA.

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