Living on the edge: Forest cover threshold effect on endangered maned sloth occurrence in Atlantic Forest

Paloma Marques Santos[a,b,c,*], Larissa Lynn Bailey[d], Milton Cezar Ribeiro[b],
Adriano Garcia Chiarello[e], Adriano Pereira Paglia[a]

[a] *Laboratório de Ecologia e Conservação, Departamento de Biologia Geral, Instituto de Ciências Biológicas, Universidade Federal de Minas Gerais, Av. Presidente Antônio Carlos, 6627, Belo Horizonte, 31270-901, MG, Brazil*
[b] *Laboratório de Ecologia Espacial e Conservação (LEEC), Departamento de Ecologia, Universidade Estadual Paulista - UNESP, Av. 24 A, 1515, Rio Claro, 13506-900, SP, Brazil*
[c] *Instituto de Pesquisa e Conservação de Tamanduás no Brasil, Conjunto Porto das Barcas, sala 130, Parnaíba, 64207-750, PI, Brazil*
[d] *Colorado State University, Department of Fish, Wildlife, and Conservation Biology, 80523-1474, Fort Collins, CO, USA*
[e] *Departamento de Biologia, Faculdade de Filosofia, Ciências e Letras de Ribeirão Preto, Universidade de São Paulo, Av. Bandeirantes, 3900, Ribeirão Preto, 14040-901, SP, Brazil*

ARTICLE INFO

ABSTRACT

Habitat loss and the isolation of remaining habitats are undoubtedly the two greatest threats to biodiversity conservation, especially for the maned sloth, due to its ecological restrictions. In this study, we identified a critical threshold of forest cover for maned sloth occurrence and explored the effects of other local and landscape variables. We sampled 68 sites, where we searched for the maned sloth and collected local habitat variables. We calculated the percentage of forest cover and open areas, assessing the appropriated scale through model selection. We used occupancy models and model selection methods to identify the threshold and assess occupancy and detection probabilities. The occupancy probability of the maned sloth is 0.97, but it decreases abruptly at 35% of forest cover, reaching zero in areas with less than 20% of forest cover. The two landscape variables are the most important predictors of sloth occupancy, based on the cumulative weight of evidence, were: Forest cover (78%) and Open areas cover (46%); the latter influencing negatively maned sloth occupancy. This is the first attempt to identify the habitat requirements of the threatened maned sloth in a fragmented area using landscape and local variables. Our results imply that conservation of maned sloth will benefit from an increase in the amount of native forest at the landscape scale. Given difficulties in the creation of new public protected areas, this improvement could be achieved via the recovery of areas located in private properties that are protected by the Brazilian Forest Code.

1. Introduction

Human-dominated areas have expanded throughout the world and are primarily responsible for increased deforestation (Leblois et al., 2017). Habitat fragmentation and loss leads to patch size reduction, increases habitat isolation and edge effects, and have several negative consequences for biodiversity (Fahrig, 2003). Worldwide, most of Earth's native ecosystems are within 1 km of the habitat edge (Haddad et al., 2015), embedded in anthropogenic matrix that is often inhospitable to biodiversity (Castellón and Sieving, 2006). These anthropogenic matrices can act as a filter, restricting dynamic processes such as dispersion and movement (Moraes et al., 2018; Morato et al., 2018),

gene flow (Dixo et al., 2009), and pollination (Pavageau et al., 2017). Additionally, deforested areas have limited shelter or refuge for native fauna, exposing species to opportunist predation by domestic dogs (Doherty et al., 2017), poaching (Ferreguetti et al., 2018) and roadkill (Ascensao et al., 2017; Assis et al., 2019). All these threats are commonly related with human activities in non-habitat matrix areas (Doherty et al., 2017; Ferreguetti et al., 2018).

Among the various attributes intrinsic to anthropized environments, the amount of remaining habitat is perhaps one of the most influential in the dynamics of populations and communities (Fahrig, 2013). When habitat loss increases and reaches a critical threshold, small modifications in the remaining habitat can produce more severe changes in
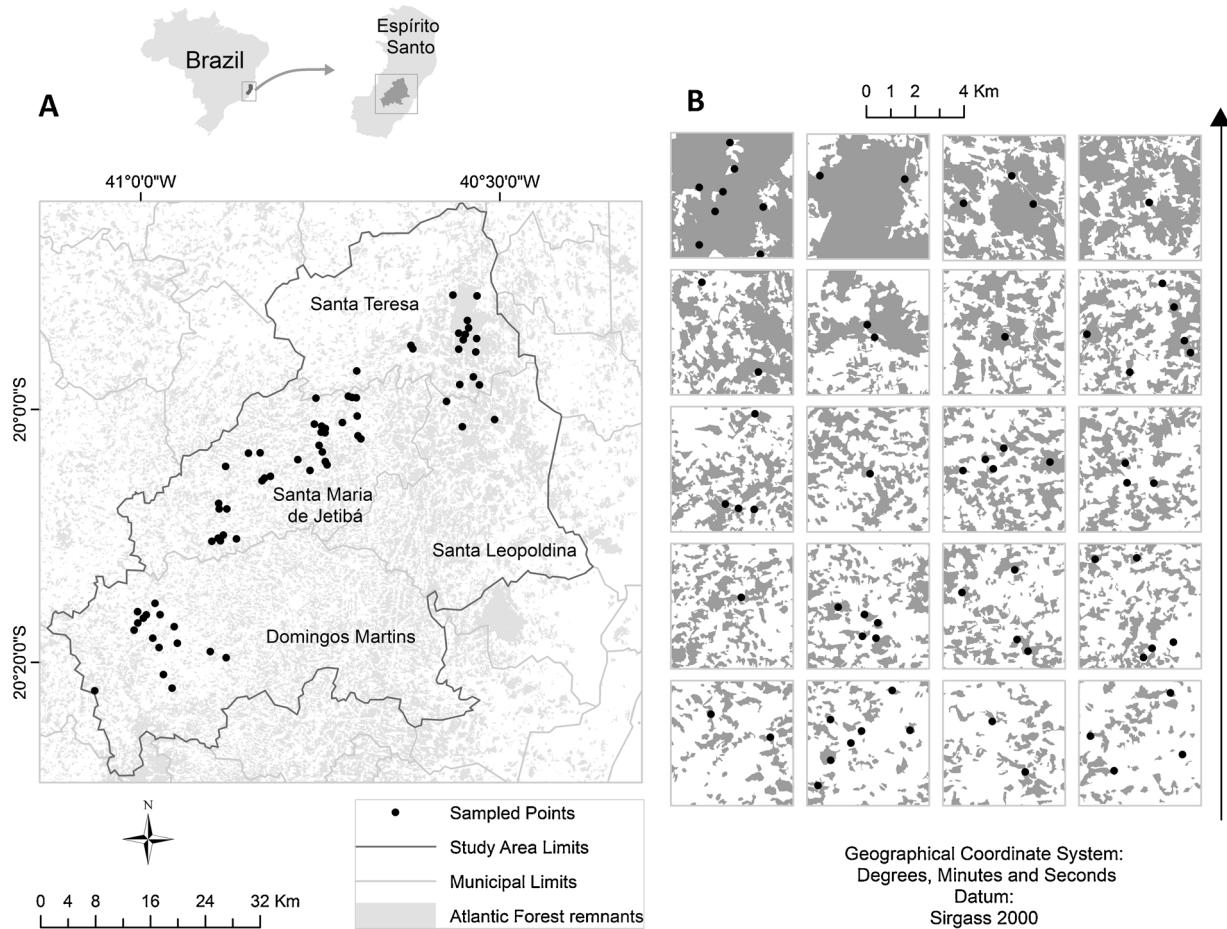

Fig. 1. Study area with the location of the 68 sampled points for recording maned sloth within Atlantic Forest remnants, Espírito Santo, Brazil (A). Example of the forest cover gradient throughout the region, in which the sampled points are inserted (B). The right-side arrow on Panel B indicated the north-south forest cover gradient arrangement.

biological responses (With and King, 1999; Huggett, 2005). Theory predicts that structural and functional connectivity suffers when the amount of remaining habitat drops below this critical threshold, and a species distribution becomes more dependent on patch size and isolation (Lande, 1987; Fahrig, 2003). The threshold value will vary among species, following the species' habitat requirements (Muylaert et al., 2016), patch size (Magioli et al., 2015), matrix composition (Boesing et al., 2018), and biome (Melo et al., 2018).

As the proportion of suitable habitat increases (i.e. above a given threshold), other variables may drive the ecological dynamics, besides the landscape context, such as habitat quality. The local habitat structure and composition plays an important role in occurrence and persistence of animal communities (August, 1983; Santos et al., 2016). Increasing habitat complexity creates a safe and dynamic environment, providing different resources and conditions necessary for reproduction (Shenbrot et al., 2002) and foraging success (Jacob et al., 2017). In addition, a more complex habitat structure may increase the alpha and beta diversities by providing suitable habitats for a higher number of species (Camargo et al., 2018).

The Atlantic Forest is a biodiversity hotspot (Zachos and Habel, 2011) and one of the most threatened biomes in the world and its area has been reduced to 28% of its original cover (Rezende et al., 2018). Most remaining forest patches are smaller than 50 ha and about half of the remaining forest is within 100 m of any forest edge (Ribeiro et al., 2009). Landscape composition and configuration pose a threat to several species, including the endemic maned sloth (*Bradypus torquatus*; Bradypodidae). According to Santos et al. (2019a), in a data paper that compiled the largest data set on abundance and occurrence of

Neotropical Xenarthrans, maned sloth (with 638 records) are distributed along Atlantic Forest cost from Sergipe to Rio Janeiro, Brazil. The species inhabits dense ombrophylous forests (Hirsch and Chiarello, 2012) and feeds mainly on leaves and shoots (Montgomery and Sunquist, 1975; Chiarello, 1998b). The species rarely descends to the ground, moving mostly through the forest canopy (Chiarello, 1998a). Due to its strictly arboreal habit and its low capacity to move in non-forested areas, the species is highly affected by the habitat loss (Chiarello and Moraes-Barros, 2014). Its restricted distribution and the dependence on forest led the International Union for Conservation of Nature (IUCN) to classify the maned sloth as Vulnerable (VU) in 2014 (Chiarello and Moraes-Barros, 2014).

There is strong evidence that habitat loss and fragmentation negatively influence maned sloth occurrence (Chiarello and Moraes-Barros, 2014). Intuitively, increasing the connection between fragments would be a logical strategy to improve the conservation of the species. Nevertheless, what exactly does "improve connection" mean for the maned sloth? What are the minimum habitat amount requirements to maintain this species? Is there a critical threshold, below which landscape variables would most likely influence the species occurrence? These questions are relevant to decision-makers responsible for applying resources to direct conservation efforts; these efforts are led by the National Center for Research and Conservation of Brazilian Primates (CPB) and linked to Chico Mendes Institute for Biodiversity Conservation (ICMBio), a Brazilian federal agency responsible for the National Action Plans.

In previously study, Santos et al. (2016) found a strong positive relationship between local variables (e.g., canopy height and

Table 1

Local and landscape level variables thought to influence maned sloth occupancy probability (Ψ) or detection probability (p). Hypothesized positive (+) or negative (-) effects are given for each parameter.

Variable	Level	Ecological Importance	Parameter	Effect on Ψ	Effect on p
Forest cover (%)	Landscape	Breeding Foraging success Shelter	Ψ	+	
Open areas cover (%)	Landscape	Dispersal Exposure to threats	Ψ	–	
Important trees (Proportion of locations)	Local (5 m)	Food source	Ψ/p	+	+
Mean canopy height (m)	Local (5 m)	Shelter Thermoregulation	Ψ/p	+	–

proportion of important trees) and maned sloth occurrence – these local variables were more important than factors related to habitat fragmentation (e.g., patch size and isolation metrics). However, the studied area encompassed a region with > 36% of forest cover, a level that exceeds forest thresholds found in other ecosystems with different taxa (thresholds ∼30 – 35% forest cover; Andrén, 1994; Estavillo et al., 2013). Motivated by our previous experience, here, we expanded the study region to encompass a broader area covering the full gradient of available habitat (5–100% forest cover) to identify and assess larger-scale variables for maned sloth occurrence. In this study, we aimed to better understand what affects maned sloth occupancy probability at the landscape level and in a local context to improve the subsidies for landscape and local management, respectively. We also contribute to the understanding of maned sloth habitat requirements, which are still unclear. We hypothesize there is a threshold relationship between the forest cover (%) and maned sloth occurrence: above this threshold maned sloth occurrence should be high and governed primarily by local variables and below this threshold maned sloth occupancy should decline rapidly regardless of local variables.

2. Material and methods

2.1. Study area

We conducted this study in the mountainous region of Espírito Santo State in southeast Brazil (Fig. 1), which historically consisted entirely of the Atlantic Forest (Muylaert et al., 2018). The study region encompasses an area of approximately 187,000 ha (Fig. 1), composed primarily of dense lower-montane and montane ombrophylous forests (Instituto Brasileiro de Geografia e Estatística, 2012). The elevation ranges from 600 to 1200 m.a.s.l. and temperatures are mild throughout the year (mean annual temperature = 22.1 °C). The annual mean precipitation is 1232 mm, with most rain concentrated between November and March (INCAPER et al., 2013a, 2013b, 2013c, 2013d).

The study region has 34% of native forests, concentrated (> 85%) in the northeast, where the largest protected areas are located, while the southern part has less than 20% of forest cover (Fig. 1). Most forest areas are privately owned (97.2%), mainly by descendents from European settlers, who arrived at the region in the early 19th century, and established small agricultural properties and a family farming system – which remains the economic base today (INCAPER et al., 2013a, 2013b, 2013c, 2013d).

2.2. Sampling design and covariates selection

We selected 68 sampling points (i.e., sites) using GrassGis 7.4.1 software (GRASS Development Team, 2018) across a range of forest cover (5–100%) within the study region. Specifically, we select 19 sites on northeast (higher percentage of forest cover), 16 sites in the south (lower percentage of forest cover), and 33 sites from the central Santa Maria de Jetibá region (medium percentage of forest cover). To minimize spatial autocorrelation, sites were at least 500 m away from each other, a distance that is large enough to accommodate the home ranges

of multiple individuals – the average sloth home range was estimated at 4.5 ha in previous studies conducted in this same area (Chiarello, 1998a). Sloth surveys were conducted during the dry season to avoid effects of rain on species detection. Specifically, we surveyed 33 sites in the central Santa Maria de Jetibá region from April to August in 2014 (Santos et al., 2016), and 35 different sites from April to October in 2017 and from March to July in 2018. The sites surveyed in 2014 were not surveyed in 2017/2018 and vice-versa. Surveys within a given dry season were spaced at approximately 1-month intervals such that each site was surveyed six times. During each survey, we established two perpendicular 200-m transects, forming a cross (Figure A.1). We walked the resulting four 100-m long segments at a speed of about 0.24 km/h, searching the canopies for maned sloths. To maximize detection, we walked the transects between 7 a.m. - 5 pm, when the species is most active (Chiarello, 1998a; Giné et al., 2015). We measured two local variables, canopy height (m) and the proportion of important trees (hereafter important trees) at five locations, including the center point (intersection of the two perpendicular transects) and at the halfway point of each 100-m transects (Figure A.1). At each location, we measured the canopy height with a rangefinder (Bushnell model Sport 850, 4 x 20 mm) and we recorded the presence/absence of important tree species (*Micropholis venulosa*, *Cupania furfuracee*, *Crecopia* spp., and *Ficus* spp.) within a 5 m radius of each location. Collectively, these species represent more than 40% of the maned sloth diet (Chiarello, 1998b). We combined the measurements at the five locations for each site to determine the site-specific mean canopy height and the proportion of locations with at least one of the important trees.

We also calculated two compositional landscape structure variables that we felt may influence maned sloth probability of occurrence: forest cover (%) and open areas cover (%). We expected each covariate of interest to influence maned sloth ecology and biology differently (Table 1). For example, forest cover is important to reproduction, foraging success, and other aspects of maned sloth ecology and behavior. Therefore, we expected a positive relationship between the forest cover and maned sloth occurrence. Conversely, open areas cover may deter or hamper the species' dispersal, since moving through these areas may expose sloths to predation, starvation, temperature stress, and other hazards. As the open areas cover increases, we expect maned sloth occupancy to decrease. At the local level, important trees are a necessary food source and consequently, maned sloth should occur more frequently in areas with a high proportion of those trees, leading to higher occupancy and detection probabilities. The relationship with detection should also be positive since it may be easier to detect the species if individuals spend more time in or around these food sources. Many trees also serve as shelter where sloths can safely rest and thermoregulate appropriately, which is usually done in the upper strata of the forest canopy. Accordingly, we expect occupancy probability to increase with increasing canopy height. However, detection is more difficult the higher the sloth is in the canopy since vegetation density also increase with canopy height, obstructing our viewing capacity.

To calculate landscape variables for each sampled site, we used a 2015 land use and land cover map with 10-m resolution, georeferenced in the UTM Projection System, SIRGAS 2000 Datum, Zone 24 s,

provided by Geobases (IEMA-ES, https://geobases.es.gov.br/). We ran the landscape analyses with Fragstats 4.2 (McGarigal et al., 2012) and considered six buffer sizes (i.e. scale of effect or spatial extension; Miguet et al., 2015) for each sampled sites: 100 m, 200 m, 300 m, 400 m, 500 m and 600 m. The usage of multiple spatial extents when evaluating the scale of effect of landscape structure influence on species occurrence, abundance or species richness is of utmost importance in landscape ecology studies (Gestich et al., 2019). Within each buffer, we calculated the percentage of forest cover and the percentage of open areas cover – the latter including existing and abandoned pasture and exposed soil.
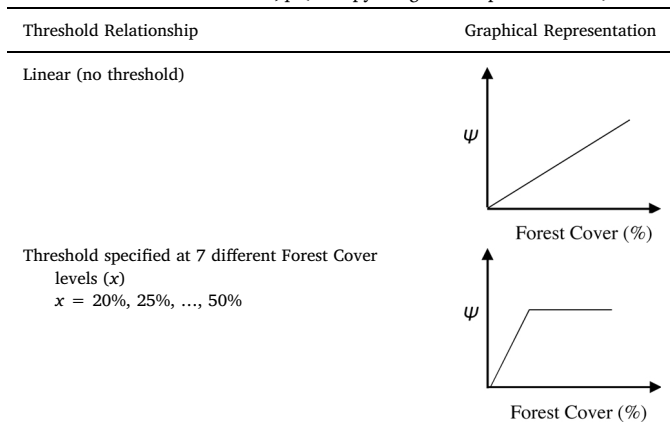
2.3. Occupancy and detection probabilities

We used a step-wise approach (four steps) to evaluate factors influencing maned sloth occupancy and detection probabilities. First, we determined the scale of effect (buffer size) for each of our landscape covariates: percentage of forest cover and percentage of open areas cover. Using a general model structure where both occupancy and detection probabilities are modeled as additive functions of local variables (important trees and canopy height), we fitted 36 models, where sloth occupancy was modeled as a linear function (on the logit scale) of all possible combinations of the six spatial extents for percent forest cover and percent open areas cover (Table A.1). We used the best-supported scale of effect for each of the landscape variables in subsequent analyses. We also performed a correlation analysis using the best-supported scale of effect for forest cover (500 m) and open area cover (600 m) and our local covariates to ensure that none of our covariates were highly correlated ($|r| < 0.6$; Table A.2).

Second, we explored if there was evidence of a threshold relationship between the probability of sloth occurrence and the percent of forest cover, using a general detection probability structure and an occupancy structure that accounted for potential variation associated with our other covariates. Specifically, we built eight occupancy structures: one corresponding to the linear relation (on the logit scale) between the sloth occupancy probability and percent forest cover and the other seven models reflected threshold effects at specified values of forest cover (e.g., 20%, 25%, 30%,…, 50% forest cover; Tables 2 and A.4). For example, a model with a specified threshold of 30% forest

Table 2

Graphical representation for different threshold relations between maned sloth occupancy probability (Ψ) and forest cover (%) in Atlantic Forest, Espírito Santo, Brazil. Eight threshold relationships were fit to the data including a linear relationship (no threshold) and seven threshold models where the specified threshold value (x) varied among models and ranged from $x = 20\%$ forest cover to $x = 50\%$ forest cover, vary by increments of 5%. All threshold models had eight parameters, allowing sloth occupancy probability to vary with other considered variables, Ψ (Open areas cover + Important Trees + Canopy Height + *Threshold Relationship*) and detection probability was modeled as function of both local variables, p (Canopy Height + Important Trees).

Threshold Relationship	Graphical Representation
Linear (no threshold)	
Threshold specified at 7 different Forest Cover levels (x) $x = 20\%, 25\%, …, 50\%$	

Table 3

Model selection results for supported models of: (A) thresholds effects sloth occupancy and forest cover (%), (B) detection probability structure and (C) occupancy probability structure. The general detection structure, p (Canopy Height + ImpTrees), was used for all threshold models (A). The best-supported forest cover threshold value and additional covariates were included in the general occupancy structure (Ψ (Open areas cover + ImpTrees + Canopy + 35% Forest Cover Threshold) used to evaluate the four detection probability structures (B). To reduce potential bias, we used the supported p (Canopy Height) detection structure when evaluating factors influencing sloth occupancy (C). Measurements of model fit (-2Log(L)), number of parameters, and Quasi-AICc values are comparable across all model evaluation steps (A, B, and C). Model weights and difference in QAICc (ΔQAICc) are calculated within steps only to help identify the best supported model structure within a given step. The bolded model represents the most parsimonious model in the candidate model set and the constant occupancy structure is given as a reference. ImpTrees = Important Trees.

Model	QAICc	ΔQAICc	w	-2Log(L)	K
(A) Threshold Relationship: Ψ (Open areas cover + ImpTrees + Canopy Height + *Threshold Relationship*)					
35% Forest Cover Threshold	**117.32**	**0**	**0.21**	**250.07**	**8**
30% Forest Cover Threshold	117.70	0.37	0.18	251.02	8
40% Forest Cover Threshold	118.09	0.77	0.15	252.01	8
(B) Detection probability structure					
p (.) – constant detection structure	114.30	0	0.44	255.22	6
p (Canopy Height)	**114.98**	**0.68**	**0.32**	**250.64**	**7**
p (ImpTrees)	116.56	2.26	0.14	254.63	7
p (Canopy Height + ImpTrees)	117.33	3.03	0.01	250.07	8
(C) Occupancy structure					
Ψ (35% Forest Cover Threshold)	**110.24**	**0**	**0.20**	**255.92**	**4**
Ψ (35% Forest Cover Threshold + Open area cover)	111.09	0.86	0.13	252.11	5
Ψ (35% Forest Cover Threshold + ImpTrees)	111.35	1.12	0.12	252.748	5
Ψ (.) – constant occupancy structure	115.01	4.77	0.02	273.98	3

cover suggests that sloth occupancy starts to decline when the forest cover falls below 30%, but sloth occupancy is approximately equal at sites above the threshold (see Table 2 for a graphical representation of these relationships). We modelled thresholds from 20% to 50% forest cover (increasing in 5% increments) and used a model selection criteria (AICc) to determine which threshold level was best supported by our data (Table A.4).

Third, using the best supported threshold occupancy structure, we explored local factors that could influence sloth detection probability. Specifically, we fitted four detection probability structures that included all additive combinations of canopy height and important tree (Table 3). Finally, we considered occupancy structures to explore additive (16 models) and interactive effects (6 models) of our local and landscape variables, fitting models with up to 7 parameters to avoid the overparameterization. We explored interactive relationships between forest cover and the other occupancy covariates because the effect of some variables (e.g., important trees and open areas cover) may change as the forest cover increases.

All models were fitted using the single-season occupancy model (MacKenzie et al., 2006) incorporated in program MARK (White and Burnham, 1999). Using our global model, we conducted a goodness-of-fit test and estimated overdispersion ($\hat{c}$) using the parametric bootstrap procedure (MacKenzie and Bailey, 2004) implemented in program PRESENCE (software version 2.12.20; Hines, 2006). We evaluated model selection using second-order Akaike Information Criterion (AICc) or Quasi-Akaike Information Criterion (QAICc; Burnham and Anderson, 2002), selecting only those models with ΔAICc < 2 (Mackenzie et al., 2006). In addition, we calculated cumulative model weights (w_+) for each explanatory variable in our balanced set of models associated with each of the four steps described above (Burnham and Anderson, 2002).

Then, we used estimated effects from our best-supported model to generate predict maps for the maned sloth distribution at the study area.

3. Results

We detected the maned sloth 53 times at 32 out 68 sampled sites (naïve occupancy estimate = 47%). Among the sites with sloth detections, 53% had only one detection, 34% had two detections, and 6% had three or four detections. The forest cover around the sampled sites range from 5% to 100%, open areas cover ranges from 0 to 69%, the proportion of important trees range from 0 to 1 and canopy height range from 8.9 to 30.5 m (Table A.3).

The evaluation of the scale of effect (buffer size) for our two landscape variables suggested that a model that included forest cover at 500 m and open area cover at 600 m was approximately twice as likely as any other model in candidate set (Table A.1). These scales of effects were used in all subsequent modeling. The parametric bootstrap goodness of fit revealed some evidence of overdispersion ($\hat{c} = 2.59$), so we used Quasi-AICc (Burnham and Anderson, 2002) for model selection procedures. Our best-supported forest threshold relationship suggested that maned sloth occupancy declined when forest cover was below 30–35% (Tables 3A, A.4).We found some evidence that detection probability was positively affected by canopy height ($w_+ = 0.32$; $\hat{\beta} = 0.09$, $SE(\hat{\beta}) = 0.07$; Fig. 2), but a model with constant detection probability was more parsimonious (model weight = 0.44, Table 3B). To ensure less bias in our occupancy estimates, we used the more general structure where detection could vary among sites with different canopy height in our evaluation of factors influencing sloth occupancy.

Among competing occupancy models, three had the ΔQAICc < 2 (Tables 3C, A.5). Our best-supported model suggests that sloth occurrence is extremely high (near 1.0) when there is over 35% forest cover surrounding a site, but sloth occurrence drops rapidly when forest cover is below this threshold (Fig. 2). Our data also suggested that sloths do
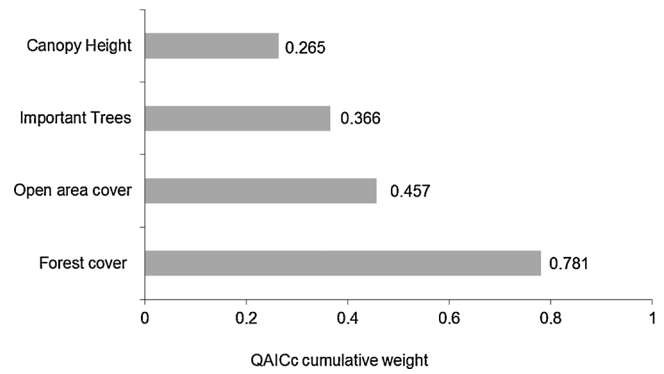

Fig. 3. Cumulative weight of evidence (QAICc cumulative weight) for each covariate used to model maned sloth occupancy within Atlantic Forest, Espírito Santo, Brazil.

not occur at sites with ≤ 20% of forest cover (Fig. 2). As expected, the proportion of important trees showed a positive relationship with occupancy probability and the percentage of open areas cover had a negative influence on sloth occurrence (Fig. 2). We found no evidence of interactive relationship between the forest cover and the other three variables (Table A.5), possibly due to our distribution of sampling sites. Landscape variables have the most pronounced importance: forest cover had the highest cumulative weight ($w_+ = 0.78$), followed by the open area cover ($w_+ = 0.46$). The two local variables had the lowest cumulative weights $w_+ < 0.40$ (Fig. 3).

Our predictive maps generated from the top three occupancy models show the maned sloth occurring with higher probabilities in a relatively wide band that stretches approximately north-southward in the study region (Fig. 4). In all three predictions, the maned sloth has a high occupancy probability in the northeast and central part of the study area. Conversely, the sloth is almost absent in the areas in the
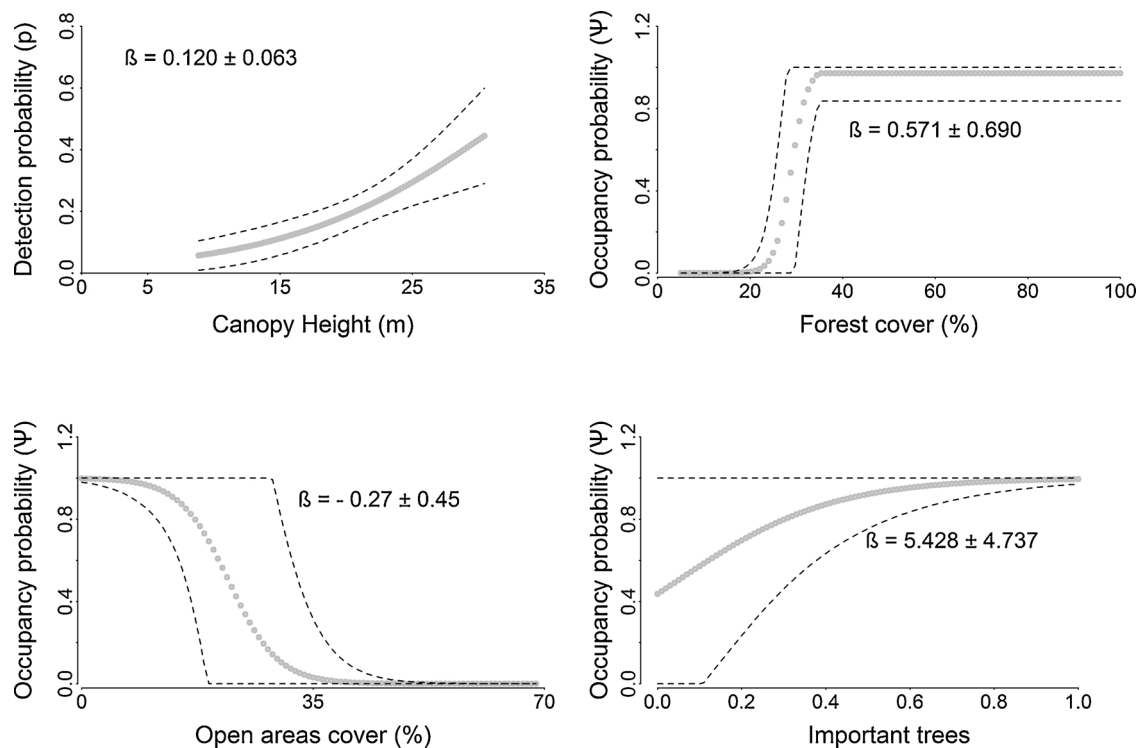

Fig. 2. Maned sloth detection and occupancy probabilities at 68 sampled sites within Atlantic Forest, Espírito Santo, Brazil, predicted by the three best-supported models. The slope parameters (betas) estimated for the covariates "forest cover", "open areas cover" (second ranked model) and "important trees" (third ranked model) are also shown. The gray line indicates the estimated covariate relationship and the black dash indicate the standard error. The relationships between open areas and important trees are graphed using the mean forest cover (%).
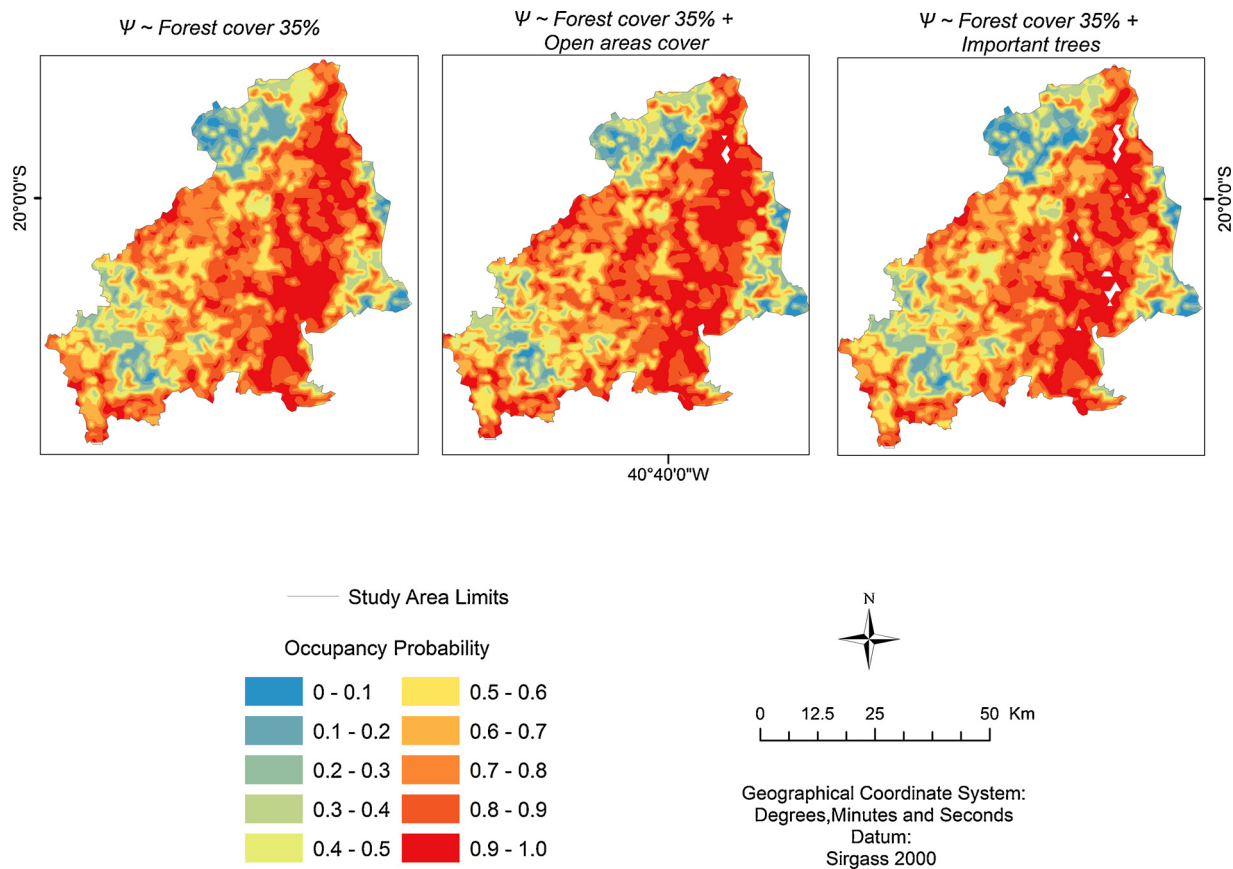
Fig. 4. Predicted estimates of maned sloth occupancy probability in the study area within Atlantic Forest, Espírito Santo, Brazil; the three best-supported occupancy models were used.

south, north and east edge – which corresponds to the areas with low forest cover.

4. Discussion

We found a threshold effect for the occupancy probability of the threatened and cryptic maned sloth. In contrast to previous study (Santos et al., 2016), our results suggest that both forest cover and open areas cover strongly affect maned sloth occurrence across forest fragmentation gradients. The effect of local variables on detection probability at occupied sites was similar to our previous study, with canopy height positively influencing the maned sloth's detection. This positive relationship was contrary to our expectation and may be related to the higher abundance of the maned sloth in taller forests which, apparently, overcompensates the difficulties in finding sloths high up in the canopy.

4.1. Thresholds effects on occupancy probability

The best-supported model predicts that areas with more than 35% of forest cover (65% of the study area) are likely to support maned sloth ($\hat{\psi}$ = 0.97, SE ($\hat{\psi}$) = 0.13), but occupancy estimates decrease abruptly in areas below that threshold (Fig. 2), which corresponds to 35% of the study area (Fig. 4). Other habitat specialist groups exhibit similar responses to changes in habitat amount, including small to medium-sized terrestrial mammals (Estavillo et al., 2013; Ochoa-Quintero et al., 2015), woody plants (Lima and Mariano-Neto, 2014), and birds (Zuckerberg and Porter, 2010). We also observed the absence of the maned sloth in areas with less than 20% of forest cover (15% of the study area; Fig.4). Forest loss in the region is already severe (originally 100% forest cover) and additional forest loss will increase the isolation of remaining forest patches and decrease patch size. These processes are

likely to lead to the local extinction of sloth in some patches and reduce sloth movement between patches making recolonization difficult.

As a forest specialist, habitat loss represents one of the major threats for the maned sloth. Landscapes with high concentration of natural habitats have smaller inter-patch distances and large numbers of patches (number of patches peaks at 30%, see Gustafson and Parker, 1992). This configuration allows the maned sloth – a strictly arboreal species – to move easily between forest patches. As the forest cover decreases, patch isolation and matrix resistance increase, followed by a decrease in patch size (Andrén, 1994; Swift and Hannon, 2010). Isolated and small fragments might no longer support viable populations of maned sloth. In general, community composition may exhibit a shift (Pardini et al., 2010; Banks-Leite et al., 2014; Magioli et al., 2015), holding mainly generalists species, since they can exploit different types of habitat to complement their diet and behavior or resource needs (Estavillo et al., 2013; Lima and Mariano-Neto, 2014).

4.2. Interactions between forest cover and local/landscape levels

Contrary to our expectations, we found no evidence of interaction between the forest cover and the local variables, though our findings may be influenced by our limited number of sites with forest cover values between 20–35%, where sloth occupancy estimates are greater than 0, but less than 1 (Fig. 2). Forest structure and composition play an unquestionable role in sustaining high biodiversity (August, 1983). Habitat loss may cause important changes in the habitat structure of remaining patches, with an increase of light demanding arboreal species (early successional), and canopy openness, and a decrease of overall basal area (Rocha-Santos et al., 2016; Benchimol et al., 2017). Species richness may decrease within patches (Andrade et al., 2015; Magioli et al., 2015), and the compositional dissimilarity of plants

between fragments may increase (Arroyo-Rodríguez et al., 2013; Benchimol et al., 2017). This simplification of habitat composition and structure can limit food resource for the maned sloth, since in general, the sloths have a very restrictive diet, feeding on a few arboreal species (Chiarello, 1998b; Montgomery and Sunquist, 1975), including some shade tolerant species (Chiarello, 1998b). The reduction of canopy height and increase of canopy openness can also influence sloths, since they spend most of their time on high canopy strata resting, feeding, breeding, and executing thermoregulatory behavior (Montgomery and Sunquist, 1975; Pauli et al., 2016; Garcés-Restrepo et al., 2017).

Our second-best model suggests that sloth occupancy is influenced by both the open areas cover and forest cover, with a strong negative effect of open areas (Fig. 2). Therefore, open areas may limit maned sloth movements, and the species may avoid it – a pattern also found previously (Falconi et al., 2015). Sloths can survive in small fragments of forest surrounded by an intermediate forest cover (20–35%) due to their relatively small home range (Chiarello, 1998b; Falconi et al., 2015). They can exploit living fences (e.g. narrow linear strips of planted trees; León and Harvey, 2006), isolated trees, forestry systems, and agriculture crops if these habitat patches provide food resources or are located between potentially accessible forest fragments (Cassano et al., 2011; Peery and Pauli, 2014; Vaughan et al., 2007). This is also true for other arboreal mammal, such as *Alouatta palliata mexicana* (Asensio et al., 2009), *Colobus anglensis palliatus* (Anderson et al., 2007), and *Callicebus personatus* found in our study area (Pers. Observation). The use of living fences, isolated trees, forestry systems and agriculture crops – habitats with poor resources and conditions – by animals that are considered strictly arboreal provides evidence of the species adaptive capacity to maximize resource consumption in more anthropogenic landscapes due to the lower quality of existing forest fragments. However, the use of open areas may represent a severe predation hazard since the sloths, and other arboreal species, are frequently exposed to attacks by domestic and feral dogs (Vaughan et al., 2007; Oliveira et al., 2008; Garcés-Restrepo et al., 2018). Similarly, since disturbed forests are of low quality, the sloths would be foraging on a lower and more open canopy, becoming more exposed to predation by ocelot (*Leopardus pardalis* - Delibes et al., 2011) and tayra (*Eira Barbara* - Sáenz-Bolaños et al., 2018). In addition, the energy expenditure to cross larger open areas may be higher than its putative resource gains, since the sloth anatomy is not adapted to movements on the ground (Goffart, 1971).

4.3. Implications for conservation

We provide the first assessment of the maned sloth habitat requirements in a broader landscape context, representing the full spectrum of forest cover gradient, and with a multi-scale perspective covering a wide range of scale of effects. This understanding can lead to better targeted conservation actions, including a more accurate assessment of the species conservation status and threats. The use of thresholds in conservation and management actions might be debated, mainly because the threshold value may exclude species that require more contiguous habitat (Johnson, 2013; van der Hoek et al., 2015). Additionally, when habitat loss is severe, biodiversity recovery may require different conservation approaches and the previous conservation state may never be obtained (*Hysterisis*; Johnson, 2013; van der Hoek et al., 2015). Still, maintaining 30%–35% of forest cover in an area can be important for a wide diversity of species and may conserve several ecological process essential to ecosystem function (Martensen et al., 2012; Banks-Leite et al., 2014; Muylaert et al., 2016).

To improve maned sloth conservation, our results support the need for an increase in forest cover, which will also increase habitat connectivity, by: 1) restoring and increasing riparian vegetation and 2) increasing matrix permeability with agroforestry systems and living fences. The Atlantic Forest has lost much of its riparian vegetation, and if this vegetation could be restored, the forest cover could increase to 30% (Rezende et al., 2018), approaching the threshold level found in

our study. Sloths typically use riparian forest for foraging and dispersal (Ramirez et al., 2011; Garcés-Restrepo et al., 2018) and an early study observed maned sloths using swamp forests in Poço das Antas Reserve (Pinder, 1985). These areas are also useful for birds (Tremblay and St. Clair, 2011), butterflies (van Halder et al., 2015), and bats (Wordley et al., 2015; Muylaert et al., 2016). Additionally, riparian vegetation has a positive impact on water quality by reducing the nutrient and sediment load in streams (Dosskey et al., 2010).

Still, conserving and restoring riparian vegetation may not be enough to ensure the conservation of maned sloth. According to the Brazilian Native Vegetation Protection Law (Federal Law 12,727/2012), 20% of forest areas in rural properties must be set aside as legal forest reserves. However, this value is far below our threshold of 35%, and we found that maned sloth is unlikely to exist in areas with less than 20% of forest cover (Fig. 2). Thereby, we reinforce that areas with less than 20% forest cover are unlikely to sustain viable populations of sloths. Additionally, the current law incorporates riparian vegetation in the calculation of the legal reserves, resulting in future deficits in the overall vegetation (Soares-filho et al., 2014), including our study area where 97.3% of forest cover are privately owned. To guarantee more consistent conservation, the restoration of riparian forest must be followed by forest reforestation in other areas.

By mixing native vegetation with agricultural production, the agroforestry systems may be a sustainable way of reducing the contrast between the forest cover and the matrix, along with increasing forest area (Schroth and Harvey, 2007; Santos et al., 2019b). When combined with other forested areas, these systems can offer more suitable habitat for maned sloths than the traditional agricultural system (Cassano et al., 2011). Moreover, providing different trees in the form of living fences yield new resource areas, resting refugia, and can act as movement corridors between habitat patches for the sloths, enhancing the landscape connectivity and reducing the pressure to traverse the anthropogenic open matrix (Castellón and Sieving, 2006; León and Harvey, 2006). Together, the restoration and increase of riparian vegetation and the creation of living fences and biodiverse agroforestry system may provide useful corridors and habitats for the maned sloth and for a wide range of species, thus allowing a long-term biodiversity conservation.

In this paper, we found a threshold relationship between maned sloth occurrence and forest cover using an occupancy modeling framework. This framework is recommended for species with low detection probabilities, such as sloth species. Sloth occurrence declined quickly in areas with less than 35% forest cover and sloth were unlikely to occur in areas with less than 20% forest cover. Future studies aiming at understanding the potential interaction between landscape and local variables for this species may concentrate effort within this forest cover range and could provide valuable information for conservation efforts in degraded forest areas. In addition, we strongly recommend long-term monitoring of sloth in our study region – as well as in other key distribution areas of the species –incorporating movement monitoring, to estimate changes in occupancy over time and determine sloth movements and resource use within and between forest patches surrounded by different types of non-habitat and poor-habitat matrix. Such studies would make it possible to better infer how the maned sloth uses and selects their habitats in a heterogeneous and anthropogenic landscape, improving the management and conservation of this lesser known species.

Acknowledgements

References

Anderson, J., Rowcliffe, J., Cowlishaw, G., 2007. Does the matrix matter? A forest primate in a complex agricultural landscape. Biol. Conserv. 135, 212–222. https://doi.org/10.1016/j.biocon.2006.10.022.

Andrade, E.R., Jardim, J.G., Santos, B.A., Melo, F.P.L., Talora, D.C., Faria, D., Cazetta, E., 2015. Effects of habitat loss on taxonomic and phylogenetic diversity of understory Rubiaceae in Atlantic forest landscapes. For. Ecol. Manage. 349, 73–84. https://doi.org/10.1016/j.foreco.2015.03.049.

Andrén, H., 1994. Effects of habitat fragmentation on birds and mammals in landscapes with different proportions of suitable habitat : a review. Oikos 71, 355–366.

Arroyo-Rodríguez, V., Rös, M., Escobar, F., Melo, F.P.L., Santos, B.A., Tabarelli, M., Chazdon, R., 2013. Plant β-diversity in fragmented rain forests: testing floristic homogenization and differentiation hypotheses. J. Ecol. 101, 1449–1458. https://doi.org/10.1111/1365-2745.12153.

Ascensao, F., Desbiez, A.L.J., Medici, E.P., Bager, A., 2017. Spatial patterns of road mortality of medium–large mammals in Mato Grosso do Sul. Brazil. Wildl. Res. 44, 135. https://doi.org/10.1071/WR16108.

Asensio, N., Arroyo-Rodríguez, V., Dunn, J.C., Cristóbal-Azkarate, J., 2009. Conservation value of landscape supplementation for howler monkeys living in forest patches. Biotropica 41, 768–773. https://doi.org/10.1111/j.1744-7429.2009.00533.x.

Assis, J.C., Giacomini, H.C., Ribeiro, M.C., 2019. Road Permeability Index: evaluating the heterogeneous permeability of roads for wildlife crossing. Ecol. Indic. 99, 365–374. https://doi.org/10.1016/j.ecolind.2018.12.012.

August, P.V., 1983. The role of habitat complexity and heterogeneity in structuring tropical mammal communities author (s): peter V. August published by : ecological society of america stable URL. Ecology 64, 1495–1507. http://www.jstor.org/stable/1937504.

Banks-Leite, C., Pardini, R., Tambosi, L.R., Pearse, W.D., Bueno, A.A., Bruscagin, R.T., Condez, T.H., Dixo, M., Igari, A.T., Martensen, A.C., Metzger, J.P., Banks-Leite, Cristina, Pardini, Renata, Tambosi, Leandro R., Pearse, William D., Bueno, Adriana A., Bruscagin, Roberta T., Condez, Thais H., Dixo, Marianna, Igari, Alexandre T., Martensen, Alexandre C., J.P.M, 2014. Using ecological thresholds to evaluate the costs and benefits of set-asides in a biodiversity hotspot. Science (80-.) 345, 1041–1045. https://doi.org/10.1126/science.1255768.

Benchimol, M., Mariano-neto, E., Faria, D., Rocha-santos, L., Souza, M.De, Sanches, F., Custodio, D., Cazetta, E., 2017. Translating plant community responses to habitat loss into conservation practices : Forest cover matters. Biol. Conserv. 209, 499–507. https://doi.org/10.1016/j.biocon.2017.03.024.

Boesing, A.L., Nichols, E., Metzger, J.P., 2018. Biodiversity extinction thresholds are modulated by matrix type. Ecography (Cop.) 1520–1533. https://doi.org/10.1111/ecog.03365.

Burnham, K.P., Anderson, D.R., 2002. Model Selection and Multimodel Inference, second. ed. Springer UShttps://doi.org/10.1002/1521-3773(20010316)40:6 < 9823::AID-ANIE9823 > 3.3.CO;2-C.

Camargo, N.Fde, Sano, N.Y., Vieira, E.M., 2018. Forest vertical complexity affects alpha and beta diversity of small mammals. J. Mammal. 1–11. https://doi.org/10.11821/dlxb201802008.

Cassano, C.R., Kierulff, M.C.M., Chiarello, A.G., 2011. The cacao agroforests of the Brazilian Atlantic forest as habitat for the endangered maned sloth Bradypus torquatus. Mamm. Biol. - Zeitschrift für Säugetierkd. 76, 243–250. https://doi.org/10.1016/j.mambio.2010.06.008.

Castellón, T.D., Sieving, K.E., 2006. An experimental test of matrix permeability and corridor use by an endemic understory bird. Conserv. Biol. 20, 135–145. https://doi.org/10.1111/j.1523-1739.2006.00332.x.

Chiarello, A.G., 1998a. Activity budgets and ranging patterns of the Atlantic forest maned sloth Bradypus torquatus (Xenarthra: bradypodidae). J. Zool. 246, 1–10. https://doi.org/10.1111/j.1469-7998.1998.tb00126.x.

Chiarello, A.G., 1998b. Diet of the Atlantic forest maned sloth Bradypus torquatus (Xenarthra : bradypodidae). J. Zool. 246, 11–19.

Chiarello, A.G., Moraes-Barros, N., 2014. Bradypus torquatus. The IUCN Red List of Threatened Species 2014. e.T3036A47436575. http://dx.doi.org/10.2305/IUCN.UK.2014-1.RLTS.T3036A47436575.en. Downloaded on 17 September 2019. .

Delibes, M., Calzada, J., Chávez, C., Revilla, E., Ribeiro, Ba., Prado, D., Keller, C., Palomares, F., 2011. Unusual observation of an ocelot (Leopardus pardalis) eating an adult Linnaeus's two-toed sloth (Choloepus didactylus). Mamm. Biol. 76, 240–241. https://doi.org/10.1016/j.mambio.2010.08.006.

Dixo, M., Metzger, J.P., Morgante, J.S., Zamudio, K.R., 2009. Habitat fragmentation reduces genetic diversity and connectivity among toad populations in the Brazilian Atlantic Coastal Forest. Biol. Conserv. 142, 1560–1569. https://doi.org/10.1016/j.biocon.2008.11.016.

Doherty, T.S., Dickman, C.R., Glen, A.S., Newsome, T.M., Nimmo, D.G., Ritchie, E.G., Vanak, A.T., Wirsing, A.J., 2017. The global impacts of domestic dogs on threatened vertebrates. Biol. Conserv. 210, 56–59. https://doi.org/10.1016/j.biocon.2017.04.007.

Dosskey, M.G., Vidon, P., Gurwick, N.P., Allan, C.J., Duval, T.P., Lowrance, R., Michael, G., Vidon, P., Gurwick, N.P., Allan, C.J., Duval, T.P., Lowrance, R., 2010. The role of riparian vegetation in protecting and improving chemical water quality in streams 1. J. Am. Water Resour. Assoc. 46, 1–17.

Estavillo, C., Pardini, R., Da Rocha, P.L.B., 2013. Forest loss and the biodiversity threshold: an evaluation considering species habitat requirements and the use of matrix habitats. PLoS One 8, 1–10. https://doi.org/10.1371/journal.pone.0082369.

Fahrig, L., 2003. Effects of habitat fragmentation on biodiversity. Annu. Rev. Ecol. Evol. Syst. 34, 487–515. https://doi.org/10.1146/annurev.ecolsys.34.011802.132419.

Fahrig, L., 2013. Rethinking patch size and isolation effects: the habitat amount hypothesis. J. Biogeogr. 40, 1649–1663. https://doi.org/10.1111/jbi.12130.

Falconi, N., Vieira, E.M., Baumgarten, J., Faria, D., Giné, G.A.F., Press, I., 2015. The home range and multi-scale habitat selection of the threatened maned three-toed sloth (Bradypus torquatus). Mamm. Biol. - Zeitschrift für Säugetierkd. https://doi.org/10.1016/j.mambio.2015.01.009.

Ferreguetti, Á.C., Pereira-Ribeiro, J., Prevedello, J.A., Tomás, W.M., Rocha, C.F.D., Bergallo, H.G., 2018. One step ahead to predict potential poaching hotspots: modeling occupancy and detectability of poachers in a neotropical rainforest. Biol. Conserv. 227, 133–140. https://doi.org/10.1016/j.biocon.2018.09.009.

Garcés-Restrepo, M.F., Pauli, J.N., Peery, M.Z., 2018. Natal dispersal of tree sloths in a human-dominated landscape: implications for tropical biodiversity conservation. J. Appl. Ecol. 55, 2253–2262. https://doi.org/10.1111/1365-2664.13138.

Garcés-Restrepo, M.F., Peery, M.Z., Reid, B., Pauli, J.N., 2017. Individual reproductive strategies shape the mating system of tree sloths. J. Mammal. 98, 1417–1425. https://doi.org/10.1093/jmammal/gyx094.

Gestich, C.C., Arroyo-Rodríguez, V., Ribeiro, M.C., da Cunha, R.G.T., Setz, E.Z.F., 2019. Unraveling the scales of effect of landscape structure on primate species richness and density of titi monkeys (Callicebus nigrifrons). Ecol. Res. 34, 150–159. https://doi.org/10.1111/1440-1703.1009.

Giné, G.A.F., Cassano, C.R., Almeida, S.S., Faria, D., 2015. Activity budget, pattern and rhythm of maned sloths (Bradypus torquatus): responses to variations in ambient temperature. Mamm. Biol. 80, 459–467. https://doi.org/10.1016/j.mambio.2015.07.003.

Goffart, M., 1971. Function and Form in the Sloth. Pergamon Press, Oxford.

Gustafson, E.J., Parker, G.R., 1992. Relationships between landcover proportion and indices of landscape spatial pattern. Landsc. Ecol. 7, 101–110. https://doi.org/10.1007/BF02418941.

Haddad, N.M., Brudvig, L.A., Clobert, J., Davies, K.F., Gonzalez, A., Holt, R.D., Lovejoy, T.E., Sexton, J.O., Austin, M.P., Collins, C.D., Cook, W.M., Damschen, E.I., Ewers, R.M., Foster, B.L., Jenkins, C.N., King, A.J., Laurance, W.F., Levey, D.J., Margules, C.R., Melbourne, B.A., Nicholls, A.O., Orrock, J.L., Song, D.-X., Townshend, J.R., 2015. Habitat fragmentation and its lasting impact on Earth's ecosystems. Sci. Adv. 1https://doi.org/10.1126/sciadv.1500052. e1500052–e1500052.

Hines, J.E., 2006. PRESENCE – Software to Estimate Patch Occupancy and Related Parameters, Version 7.8. USGS-PWRC (10.10.2013). www.mbr-pwrc.usgs.gov/software/presence.html.

Hirsch, A., Chiarello, A.G., 2012. The endangered maned sloth Bradypus torquatus of the Brazilian Atlantic forest: a review and update of geographical distribution and habitat preferences. Mamm. Rev. 42, 35–54. https://doi.org/10.1111/j.1365-2907.2011.00188.x.

Huggett, A.J., 2005. The concept and utility of "ecological thresholds" in biodiversity conservation. Biol. Conserv. 124, 301–310. https://doi.org/10.1016/j.biocon.2005.01.037.

INCAPER, I.C., de, P.A., técnica e, E., 2013a. Programa De Assistência Técnica E Extensão Rural Proater 2011-2013: Domingos Martins. Vitória. https://doi.org/LOCAÇÃO. .

INCAPER, I.C., de, P.A., técnica e, E., 2013b. Programa De Assistência Técnica E Extensão Rural Proater 2011-2013: Santa Leopoldina. Vitória.

INCAPER, I.C., de, P.A., técnica e, E., 2013c. Programa De Assistência Técnica E Extensão Rural Proater 2011-2013: Santa Teresa. Vitória.

INCAPER, I.C., de, P.A., técnica e, E., 2013d. Programa De Assistência Técnica E Extensão Rural Proater 2011-2013: Santa Maria De Jetiba. Vitória.

Instituto Brasileiro de Geografia e Estatística, I., 2012. Manual Técnico Da Vegetação Brasileira, 1st ed. IBGE, Rio de Janeiro, pp. 0101–4234 https://doi.org/ISSN.

Jacob, S.A., Matter, S.F., Cameron, G.N., 2017. Interactive effects of vegetation and illumination on foraging behavior of white-footed mice (Peromyscus leucopus). J. Mammal. 98, 804–814. https://doi.org/10.1093/jmammal/gyx012.

Johnson, C.J., 2013. Identifying ecological thresholds for regulating human activity: effective conservation or wishful thinking? Biol. Conserv. 168, 57–65. https://doi.org/10.1016/j.biocon.2013.09.012.

Lande, R., 1987. Extinction thresholds in demographic models of territorial populations. Am. Nat. 130, 624–635.

Leblois, A., Damette, O., Wolfersberger, J., 2017. What has driven deforestation in developing countries since the 2000s? Evidence from new remote-sensing data. World Dev. 92, 82–102. https://doi.org/10.1016/j.worlddev.2016.11.012.

León, M.C., Harvey, C.A., 2006. Live fences and landscape connectivity in a neotropical agricultural landscape. Agrofor. Syst. 68, 15–26. https://doi.org/10.1007/s10457-005-5831-5.

Lima, M.M., Mariano-Neto, E., 2014. Extinction thresholds for Sapotaceae due to forest cover in Atlantic Forest landscapes. For. Ecol. Manage. 312, 260–270. https://doi.org/10.1016/j.foreco.2013.09.003.

MacKenzie, D.I., Bailey, L.L., 2004. Assessing the fit of site-occupancy models. J. Agric. Biol. Environ. Stat. 9, 300–318. https://doi.org/10.1198/108571104X3361.

Mackenzie, D.I., Nichols, J.D., Royle, J.A., Pollock, K.H., Bailey, L.L., Hines, J.E., 2006. Occupancy Estimation and Modeling - Inferring Patterns and Dynamics of Species Occurrence, first ed. Academic Press, Boston.

Magioli, M., Ribeiro, M.C., Ferraz, K.M.P.M.B., Rodrigues, M.G., 2015. Thresholds in the

relationship between functional diversity and patch size for mammals in the Brazilian Atlantic Forest. Anim. Conserv. 18, 499–511. https://doi.org/10.1111/acv.12201.

Martensen, A.C., Ribeiro, M.C., Banks-Leite, C., Prado, P.I., Metzger, J.P., 2012. Associations of forest cover, fragment area, and connectivity with neotropical understory bird species richness and abundance. Conserv. Biol. 26, 1100–1111. https://doi.org/10.1111/j.1523-1739.2012.01940.x.

Melo, I., Ochoa-quintero, J.M., Roque, F.D.O., Dalsgaard, B., 2018. A review of threshold responses of birds to landscape changes across the world. J. Field Ornithol. 1–12. https://doi.org/10.1111/jofo.12272. 0.

Miguet, P., Jackson, H.B., Jackson, N.D., Martin, A.E., Fahrig, L., 2015. What determines the spatial extent of landscape effects on species? Landsc. Ecol. 31, 1–18. https://doi.org/10.1007/s10980-015-0314-1.

Montgomery, G.G., Sunquist, M.E., 1975. Impact of sloths on Neotropical forest energy flow and nutrient cycling. Trop. Ecol. Syst. Trends Terr. Aquat. Res. 69–98. https://doi.org/10.1007/978-3-642-88533-4_7.

Moraes, A.M., Ruiz-Miranda, C.R., Galetti, P.M., Niebuhr, B.B., Alexandre, B.R., Muylaert, R.L., Grativol, A.D., Ribeiro, J.W., Ferreira, A.N., Ribeiro, M.C., 2018. Landscape resistance influences effective dispersal of endangered golden lion tamarins within the Atlantic Forest. Biol. Conserv. 224, 178–187. https://doi.org/10.1016/j.biocon.2018.05.023.

Morato, R.G., Connette, G.M., Stabach, J.A., De Paula, R.C., Ferraz, K.M.P.M., Kantek, D.L.Z., Miyazaki, S.S., Pereira, T.D.C., Silva, L.C., Paviolo, A., De Angelo, C., Di Bitetti, M.S., Cruz, P., Lima, F., Cullen, L., Sana, D.A., Ramalho, E.E., Carvalho, M.M., da Silva, M.X., Moraes, M.D.F., Vogliotti, A., May, J.A., Haberfeld, M., Rampim, L., Sartorello, L., Araujo, G.R., Wittemyer, G., Ribeiro, M.C., Leimgruber, P., 2018. Resource selection in an apex predator and variation in response to local landscape characteristics. Biol. Conserv. 228, 233–240. https://doi.org/10.1016/j.biocon.2018.10.022.

Muylaert, R.L., Steven, R.D., Ribeiro, M., 2016. Threshold effect of habitat loss on bat richness in cerrado-forest landscape. Ecol. Appl. 26, 1854–1867. https://doi.org/10.1890/15-1757.1.

Muylaert, R.L., Vancine, M.H., Bernardo, R., Oshima, J.E.F., Sobral-Souza, T., Tonetti, V.R., Niebuhr, B.B., Ribeiro, M.C., 2018. Uma Nota Sobre Os Limites Territoriais Da Mata Atlântica. Oecologia Aust. 22, 302–311. https://doi.org/10.4257/oeco.2018.2203.09.

Ochoa-Quintero, J.M., Gardner, T.A., Rosa, I., de Barros Ferraz, S.F., Sutherland, W.J., 2015. Thresholds of species loss in Amazonian deforestation frontier landscapes. Conserv. Biol. 29, 440–451. https://doi.org/10.1111/cobi.12446.

Oliveira, V.B.De, Linares, A.M., Corrêa, G.L.C., Chiarello, A.G., 2008. Predation on the black capuchin monkey by domestic dogs in the Parque Estadual Serra do Brigadeiro. Rev. Bras. Zool. 25, 376–378.

Pardini, R., de Bueno, A.A., Gardner, T.A., Prado, P.I., Metzger, J.P., 2010. Beyond the fragmentation threshold hypothesis: regime shifts in biodiversity across fragmented landscapes. PLoS One 5. https://doi.org/10.1371/journal.pone.0013666.

Pauli, J.N., Peery, M.Z., Fountain, E.D., Karasov, W.H., 2016. Arboreal folivores limit their energetic output, all the way to slothfulness. Am. Nat. 188, 196–204. https://doi.org/10.1086/687032.

Pavageau, C., Gaucherel, C., Garcia, C., Ghazoul, J., 2017. Nesting sites of giant honey bees modulated by landscape patterns. J. Appl. Ecol. 1–11. https://doi.org/10.1111/1365-2664.13069.

Peery, M.Z., Pauli, J.N., 2014. Shade-grown cacao supports a self-sustaining population of two-toed but not three-toed sloths. J. Appl. Ecol. 51, 162–170. https://doi.org/10.1111/1365-2664.12182.

Pinder, L., 1985. Observações preliminares sobre a preguiça de coleira (*Bradypus torquatus*) (Illiger, 1811) (Edentata, Bradypodidae). In: Universidade Estadual de Campinas (Ed.), XII Congresso Brasileiro de Zoologia: 290 - 291. Universidade Estadual de Campinas, Campinas.

Ramirez, O., Vaughan, C., Herrera, G., Guries, R., 2011. Temporal and spatial resource use by female three-toed sloths and their young in an agricultural landscape in Costa Rica. Rev. Biol. Trop. 59, 1743–1755.

Rezende, C.L., Scarano, F.R., Assad, E.D., Joly, C.A., Metzger, J.P., Strassburg, B.B.N., Tabarelli, M., Fonseca, G.A., Mittermeier, R.A., 2018. From hotspot to hopespot : An

opportunity for the Brazilian Atlantic Forest. Perspect. Ecol. Conserv. https://doi.org/10.1016/j.pecon.2018.10.002.

Ribeiro, M.C., Metzger, J.P., Martensen, A.C., Ponzoni, F.J., Hirota, M.M., 2009. The Brazilian Atlantic Forest: How much is left, and how is the remaining forest distributed? Implications for conservation. Biol. Conserv. 142, 1141–1153. https://doi.org/10.1016/j.biocon.2009.02.021.

Rocha-Santos, L., Pessoa, M.S., Cassano, C.R., Talora, D.C., Orihuela, R.L.L., Mariano-neto, E., Morante-filho, J.C., Faria, D., Cazetta, E., 2016. The shrinkage of a forest : landscape-scale deforestation leading to overall changes in local forest structure. Biol. Conserv. 196, 1–9. https://doi.org/10.1016/j.biocon.2016.01.028.

Sáenz-Bolaños, C., Montalvo, V., Carillo, Eduardo, Fuller, T., 2018. Tayra (Eira barbara) predation of a brown-throated three-toed sloth (Bradypus variegatus) in Costa Rica. Edentata 1–4.

Santos, P.M., Chiarello, A.G., Ribeiro, M.C., Ribeiro, J.W., Paglia, A.P., 2016. Local and landscape influences on the habitat occupancy of the endangered Maned Sloth Bradypus torquatus within fragmented landscapes. Mamm. Biol. - Zeitschrift für Säugetierkd. 81, 447–454. https://doi.org/10.1016/j.mambio.2016.06.003.

Santos, P.M., et al., 2019a. Neotropical Xenarthrans: a dataset of occurrence of xenarthran species in the neotropics. Ecology 100. https://doi.org/10.1002/ecy.2663.

Santos, P.Z.F., Crouzeilles, R., Sansevero, J.B.B., 2019b. Can agroforestry systems enhance biodiversity and ecosystem service provision in agricultural landscapes? A meta-analysis for the Brazilian Atlantic Forest. For. Ecol. Manage. 433, 140–145. https://doi.org/10.1016/j.foreco.2018.10.064.

Schroth, G., Harvey, Ca., 2007. Biodiversity conservation in cocoa production landscapes: An overview. Biodivers. Conserv. 16, 2237–2244. https://doi.org/10.1007/s10531-007-9195-1.

Shenbrot, G., Krasnov, B., Khokhlova, I., Demidova, T., Fielden, L., 2002. Habitat-dependent differences in architecture and microclimate of the burrows of Sundevall's jird (Meriones crassus) (Rodentia: Gerbillinae) in the Negev Desert, Israel. J. Arid Environ. 51, 265–279. https://doi.org/10.1006/jare.2001.0945.

Soares-filho, b., Rajão, R., Macedo, M., Carneiro, A., Costa, W., Coe, M., Rodrigues, H., Alencar, A., 2014. Cracking Brazil' s forest code. Science (80-.) 344, 363–364.

Swift, T.L., Hannon, S.J., 2010. Critical thresholds associated with habitat loss: a review of the concepts, evidence, and applications. Biol. Rev. 85, 35–53. https://doi.org/10.1111/j.1469-185X.2009.00093.x.

Tremblay, M.A., St. Clair, C.C., 2011. Permeability of a heterogeneous urban landscape to the movements of forest songbirds. J. Appl. Ecol. 48, 679–688. https://doi.org/10.1111/j.1365-2664.2011.01978.x.

van der Hoek, Y., Zuckerberg, B., Manne, L.L., 2015. Application of habitat thresholds in conservation: considerations, limitations, and future directions. Glob. Ecol. Conserv. 3, 736–743. https://doi.org/10.1016/j.gecco.2015.03.010.

van Halder, I., Barnagaud, J.Y., Jactel, H., Barbaro, L., 2015. Woodland habitat quality prevails over fragmentation for shaping butterfly diversity in deciduous forest remnants. For. Ecol. Manage. 357, 171–180. https://doi.org/10.1016/j.foreco.2015.08.025.

Vaughan, C., Ramírez, O., Herrera, G., Guries, R., 2007. Spatial ecology and conservation of two sloth species in a cacao landscape in limón, Costa Rica. Biodivers. Conserv. 16, 2293–2310. https://doi.org/10.1007/s10531-007-9191-5.

With, K.A., King, A.W., 1999. Extinction thresholds for species in fractal landscapes. Conserv. Biol. 13, 314–326. https://doi.org/10.1046/j.1523-1739.1999.013002314.x.

White, G., Burnham, K.P., 1999. Program MARK: survival estimation from populations of marked animals. Bird Study 46, 120–138.

Wordley, C.F.R., Sankaran, M., Mudappa, D., Altringham, J.D., 2015. Landscape scale habitat suitability modelling of bats in the Western Ghats of India: bats like something in their tea. Biol. Conserv. 191, 529–536. https://doi.org/10.1016/j.biocon.2015.08.005.

Zachos, F.E., Habel, J.C., 2011. Biodiversity Hotspots: Distribution and Protection of Conservation Priority Areas. Springer, Heidelberg [Germany]; New York.

Zuckerberg, B., Porter, W.F., 2010. Thresholds in the long-term responses of breeding birds to forest cover and fragmentation. Biol. Conserv. 143, 952–962. https://doi.org/10.1016/j.biocon.2010.01.004.