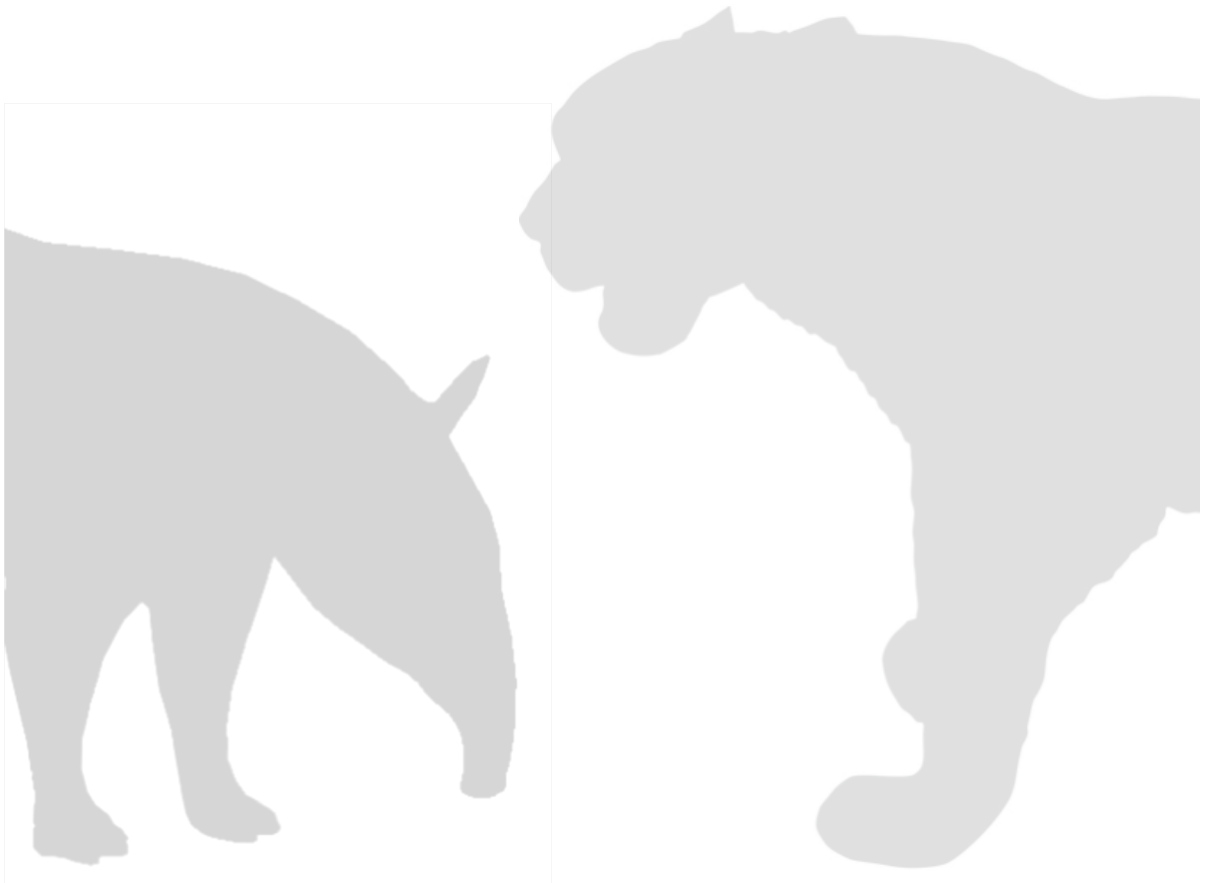# FOREST COVER AND NOT FRAGMENTATION AFFECTS COMPOSITION AND INTEGRITY OF MAMMAL COMMUNITIES IN THE NEOTROPICS

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FOREST COVER AND NOT FRAGMENTATION AFFECTS COMPOSITION AND INTEGRITY OF MAMMAL COMMUNITIES IN THE NEOTROPICS

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ABSTRACT

Deforestation poses a significant threat to tropical biodiversity and affects the assembly of species communities. Few studies aim to understand effects of deforestation on mammal communities at one location in time. We investigated the effects of a reduction in forest cover and an increase in forest fragmentation on mammal occupancies in the Chiquitano Dry Forest of Bolivia, using a multi-species occupancy model on camera trap data. Additionally, we tested how deforestation influences proportional shifts in the ratio between forest specialist and disturbance adapted species, using a linear regression. We surveyed 27 mammal species over a 5-year period during which more than half of the forest cover was lost and fragmentation increased. A reduction in forest cover had a strong negative effect on occupancy at the community level and negatively impacted more than half of the species present, including > 75% of forest specialists. In contrast, forest fragmentation had a weak effect on occupancy at the community level, promoted two species' occupancies, and did not affect any species occupancy negatively. Therefore, we conclude that the availability of forest cover is more critical for forest specialists than the degree of fragmentation. At a large forest cover, the community was dominated by forest specialists. With decreased forest cover, the proportion of forest specialists and disturbance adapted species trended towards equal. We hypothesize that other effects (i.e., secondary effects) as a result of deforestation (i.e., increased anthropogenic disturbance and microclimatic changes) are more important to determine community integrity than deforestation itself. The lag in time between deforestation and secondary effects highlights the need to understand how deforestation affects communities over time. To preserve mammalian diversity in the Chiquitano Dry Forest, our results emphasize the need to mitigate the impact of secondary effects on the mammal community.

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1 INTRODUCTION

Understanding how species communities respond to environmental change is a key objective in contemporary ecology (Ehrlén & Morris, 2015; Weiher et al., 2011). Currently, land use changes caused by anthropogenic activities are among the leading drivers of an unprecedented loss of biodiversity (Barnosky et al., 2011; Fischer & Lindenmayer, 2007; Hansen et al., 2004; Oliver & Morecroft, 2014). Most of human interferences degrade habitats and thereby affect the assembly of species communities (Hautier et al., 2015; Pardini et al., 2017). Deforestation through agricultural expansions, for example, affects the dispersal of species (Vellend, 2010, Hubbell, 2011) and the availability of niche spaces (Huston & Huston, 1994; Magurran, 2005), which are both believed to be the fundamental drivers of community assembly (Elton, 1927; Grinnell, 1917; MacArthur & Wilson, 1967; Macarthur & Levins, 1967; Mutshinda & O'Hara, 2011).

Mammal communities are particularly affected by deforestation as mammals generally require large amounts of food and space (Owen-Smith, 2002). For example, commercial logging negatively affected mammal abundances and densities in tropical forests of Africa (Brncic et al., 2015; Young et al., 2015) and Indonesia (Bernard et al., 2009). Also, a decrease in the connectivity of forest habitats caused by deforestation negatively correlated with mammalian species richness (Bovendorp et al., 2019; Romero-Muñoz et al., 2021). A decline in mammal diversity may consequently affect the functioning of an ecosystem (i.e., the effect that organisms have on the physical and chemical conditions of their environment, sensu Oliver et al., 2015) and its related services (Lacher et al., 2019). This is because mammal communities often include keystone species or ecosystem engineers that play important roles in the functioning of food web dynamics, seed dispersal, or environmental transformations (Jones et al., 1996; Power et al., 1996). Mammalian predators, for example, may influence herbivore prey population sizes (Ripple et al., 2014; Weterings et al., 2022). On their turn, mammalian herbivores can alter vegetation structure, composition, and regeneration (Hobbs, 1996; Nuñez et al., 2010; Stroh et al., 2008), which may affect other fauna such as songbirds, invertebrates, or arthropod populations (Martin et al., 2010; Rae et al., 2014).

Tropical forests harbour the most diverse communities of terrestrial mammals compared to other biomes (Blackburn & Gaston, 1997; Hillebrand, 2004), which include numerous specialized (> 75% of communities in Atlantic forests; Banks-Leite et al., 2014) and endemic species (Gentry, 1992; Myers, 1988). Specialist and endemic species are generally more

sensitive to land use changes than generalist species that can better adapt to disturbance (Jonsen & Fahrig, 1997; Morris, 2010; Newbold et al., 2020; Ramiadantsoa et al., 2018). Deforestation may therefore affect the integrity of species communities (i.e., the preservation of proportions between functional groups) at the dimension of habitat affinity (functional groups of forest specialist and disturbance-adapted species) (Andrén, 1994; Banks-Leite et al., 2014). For example, the composition of mammal communities in Atlantic forest patches shifted towards a dominance of disturbance-adapted species once the original forest cover was reduced below a threshold of 30% (Andrén, 1994; Banks-Leite et al., 2014). This in turn may negatively affect seed dispersal, pollination, maintenance of soil quality, and other faunal ecosystem services, ultimately affecting the functioning of the forest and its resilience to disturbance (Brandon, 2014; Clark, 2007; Guariguata & Balvanera, 2009; Mutoko et al., 2015).

The primary effects of deforestation that affect species communities are a reduction in forest cover and an increase in the degree of forest fragmentation (Gorczynski et al., 2021; Martensen et al., 2012; Wearn et al., 2019). A reduction in forest cover affects local conditions and resource availabilities (Huston & Huston, 1994; Magurran, 2005) and therefore alters abiotic niche spaces. This may also affect biotic niches if inter- and intraspecific species interactions are affected (e.g., through increased competition pressure) (Svenning et al., 2014; Tilman & Kareiva, 1997). Occupancy rates and relative abundances of forest specialists showed high sensitivity to forest cover (Lewis et al., 2015; Li et al., 2019; Santos et al., 2019). Fragmentation typically increases if forest cover gets reduced. Fragmentation limits dispersal abilities of species (Jayadevan et al., 2020; Lasky & Keitt, 2013; Oberosler et al., 2017) and increases the amount of forest edges that generally have unsuitable environmental conditions for forest specialists (Fischer & Lindenmayer, 2007; Pfeifer et al., 2017). Besides forest cover, fragmentation proved a strong driver to regional community assembly of particularly medium sized and large mammals (Baguette & Van Dyck, 2007; Diniz et al., 2020).

Currently, the negative effects of deforestation on biodiversity are mostly ubiquitous (Alroy, 2017; Martínez-Ramos et al., 2016). Most studies that investigate these effects compare sites with varying levels of disturbance or conservation treatments (see e.g., Banks-Leite et al., 2014; Laméris et al., 2020; Martensen et al., 2012; Wallgren et al., 2009; Wearn et al., 2019; Whitworth et al., 2019) as it allows for large sample sizes and rapid inferences (Kays et al., 2020). However, they might be biased by site-specific historic (e.g., metacommunity; see Hubbell, 2011) and abiotic (e.g., topography, soil type, macro-climate, salinity) factors that underly the present community composition in an area but are not affected by deforestation

(Pavoine & Bonsall, 2011). A comparison of sites with varying time lags between deforestation and monitoring may also fail to accurately delineate causations between community assembly and its driving factors, i.e., primary changes in the vegetation (i.e., forest cover and fragmentation) and secondary changes in the environment (i.e., microclimatic changes) or human disturbance (e.g., traffic, agriculture) (Feng et al., 2021). To control for these factors, insights of how deforestation affects site-specific communities in time require long-term monitoring of community composition at changing levels of deforestation, a design that only few studies report (Kinnaird et al., 2003; but see e.g., Ramesh et al., 2016; Sosibo et al., 2022).

Therefore, the objective in this study is to get an insight into the relationship between primary effects of deforestation at the vegetation level (forest cover and forest fragmentation) and its impact on mammal occupancies and community integrity. Occupancy rates are well suited to track responses to environmental changes at both the species and the community level (Linkie et al., 2007; Rovero et al., 2014). We used a multi-species occupancy model to estimate occupancy rates in response to deforestation using data obtained from camera traps and examined shifts in community integrity. We hypothesized that increased levels of deforestation, i.e., a reduction in forest cover and increased forest fragmentation, will (1) negatively affect occupancies of forest specialist species (Gorczynski et al., 2021; Kinnaird et al., 2003; Wearn et al., 2019), and (2) cause mammal communities to shift towards a dominance of disturbance adapted species (Banks-Leite et al., 2014).

As a case study, we investigated the Chiquitano Dry Forest (CDF) in eastern Bolivia, the largest contiguous broad-leaf dry forest in South America (Miles et al., 2006; Power et al., 1996). The Neotropics are a major hotspot of biodiversity loss due to the continuous effects of deforestation (Ibisch & Mérida, 2004; Kosydar et al., 2014; Peñaranda & Simonetti, 2015). Neotropical forests host a vast number of vertebrate species, some of which had only been described recently (Caminer et al., 2017; Jansen et al., 2011; Pansonato et al., 2020) and likely host numerous complex, yet understudied mammal communities (Anderson, 1997; Brooks et al., 2001; Meißner et al., 2023). Deforestation activities in the CDF reduced forest cover by approximately 15% between 2001 and 2006 (Killeen et al., 2006). Human-induced wildfires caused further loss of primary forest by approximately 12% in recent years (Devisscher et al., 2016; Romero-Muñoz et al., 2019). Since 2020, large scale agricultural expansions to deforest substantial parts of the CDF (Meißner et al., 2023).

2 MATERIAL & METHODS

2.1 STUDY AREA

This study was conducted in a mixed landscape of forests and agricultural land within the CDF (16°22'S 62°00'W) in Bolivia (Fig. 2). The area falls in a climatic and biogeographic transition zone between the Amazon rainforest, the Gran Chaco Dry Forest, and the Brazilian Cerrado Savanna. The average daily temperature is 24.4 °C with minimal annual variation and annual precipitation varies from 500 to 1710 mm, with a dry season from July to December (Killeen et al., 2006). The study area is located 500 meters above sea level. At the start of this study in 2017, tropical dry forests with semi-deciduous trees interspersed with woody savanna grasses and shrubs dominated the primary vegetation (ca. 80%) (Killeen et al., 2006). Agricultural activities in the area are limited to livestock, with the nearest commercial crop farming located at distance of ca. 40 km.
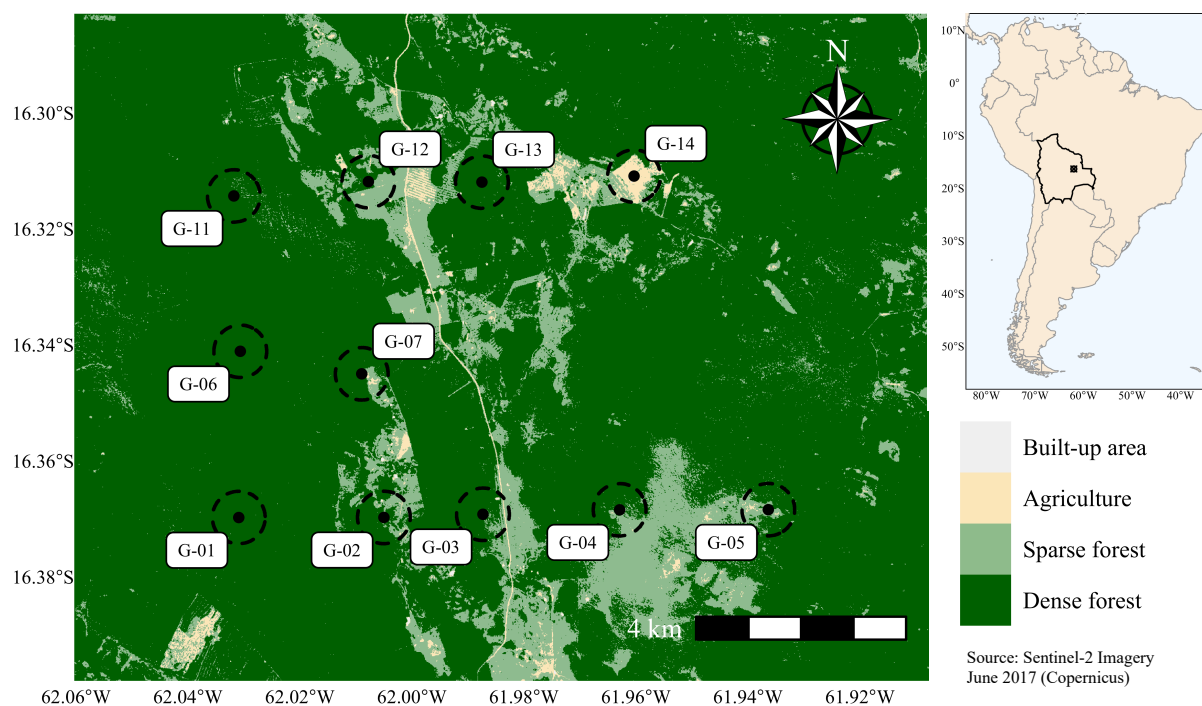
Figure 1 Location of the study area in the Chiquitano region of Bolivia indicating the arrangement of the camera trap stations (black dots) used in this study with buffer zones of a 500 m radius around each station (dotted lines).

2.2 DATA SAMPLING

2.2.1 CAMERA TRAPPING

Between January 2017 and December 2021, mammal community composition was sampled using 11 camera trap stations, each consisting of two paired cameras (Bushnell Trophy Camera

Brown Model 119437, Reconyx XR6 UltraFire, and Cuddeback G-series Double Barrel Strobe) arranged in a 10x30 km grid (Figure 1). Camera traps were placed in suitable microsites to detect medium to large sized terrestrial mammals (e.g., trails) and attached to trees at a height of 30-40 cm. The cameras were set to photo mode with a burst of three images per trigger and a minimum delay of 5 seconds between bursts. All stations contributed to 20,075 trapping nights. The stations were visited in two week intervals to change batteries and to retrieve data.

2.3   DATA COLLECTION

All subsequent steps were carried out in R version 4.1.2 (R Core Team, 2021). The packages and associated versions numbers used are referenced in appendix VI.

2.3.1   IMAGE CLASSIFICATION

We used Labelbox (Labelbox, 2021) to involve 251 citizen scientists in the image classification between September 2020 and October 2022 (project "WildLIVE! - Entdecke die wilden Tiere Boliviens", https://wildlive.sgn.one). Every image containing at least one animal was reviewed at least seven times by different citizen scientists who assigned a taxonomic classification. Of the 42,088 images containing mammals identified to the species level, 29,437 images (69.9%) were identified unanimously. For 10,314 images (24.5%) with non-unanimous labels, the identification of citizen scientists with a background in wildlife sciences (i.e., experts) was accepted. For 1,437 images (3.4%) that were not reviewed by experts, labels that reached a consensus of 70% or higher were accepted (Adam et al., 2021; Hsing et al., 2022). 900 images (2.1%) that did not yield a clear species consensus were discarded from the analysis. We retrieved the timestamp of every image taken using the Exif tool in the exifr package.

2.3.2   DEFORESTATION

We acquired two sets of four Sentinel-2 tiles (20LPH, 20LNH, 20KPG, 20KNG) at a resolution of 10 meters for each year of the study period (see appendix I) using the sen2r package. For each year, the first set of tiles were selected to represent the wet season (Jan–Jun), and the second set to represent the dry season (Jul–Dec). Tiles were chosen at a minimum difference of three months and based on the lowest mean cloud percentage. The merged tiles were cropped to a 10 km bounding box around the minimum hull-convex polygon of all camera trap stations (WGS 84, UTM 32720), allowing us to account for changes in the periphery.

2.4 DATA PREPARATION

2.4.1 MAMMAL OCCURRENCES

We defined an independent visit to a camera trap by a medium or large sized mammal (i.e., weight > 1 kg) as a visit that was at least 30 minutes after the last visit of the same species at the same camera trap station (Silveira et al., 2003). We collected mean body masses per species using literature. Every species was categorized in one of two classes of habitat affinity: forest specialist and disturbance adapted species. Distinctions were made using the framework proposed by Banks-Leite et al. (2014), who used weighted averages of habitat-related occupancies to assess sensitivity to forest cover of neotropical mammal species.

2.4.2 DEFORESTATION

We quantified forest cover and forest fragmentation using the normalized density vegetation index NDVI. NDVI values were obtained for every Sentinel-2 image and were classified into 4 habitat categories: bare soil, sparse vegetation, moderate vegetation, and dense vegetation. Cut-off values for the first and second half of the year were chosen differently to account for the effect of the dry season on the vegetation (Nath, 2014) (see appendix I for cut-off values). We calculated the proportion of each habitat category in a 500 meter radius buffer zone around camera trap stations to avoid overlap between buffer zones of different station and to reduce the effect of spatial autocorrelation (Glen et al., 2013; Lyra-Jorge et al., 2008). Forest cover was expressed as the percentage of dense vegetation (Nath, 2014; Zaitunah et al., 2018). To quantify forest fragmentation, we calculated edge length from forest cover spatial polygons. Every camera trap yielded 10 survey periods of forest cover and edge length adding up to 110 unique combinations of camera trap stations and surveys, henceforth called 'session'.

2.5 DATA ANALYSIS

2.5.1 OCCUPANCY

We modelled community and species-specific occupancy in response to forest cover and fragmentation using a custom multi-species occupancy model (i.e., global model) in a Bayesian framework, which accounts for imperfect detection probabilities. Differences in individual species' detection probabilities are a common source of bias in camera trap studies. Observed detections generally underestimate the relative abundances of rare species, inflate detection probabilities at camera trap stations that are closer to an animals home range centre, and may be influenced by proximity to e.g., waterfalls or game trails (Nichols et al., 1998; Royle, 2006;

Tobler et al., 2015). The model was fit using a JAGS vignette inside the camtrapR package. Prior to modelling, the data was explored for outliers using Cleveland plots, linearity using regression plots, multicollinearity using Pearson's correlation coefficient, and spatial independence using Moran's I test (using $\alpha = 0.05$) without finding relevant issues. Additionally, we examined the sampling effort using an extrapolated rarefaction curve in the iNEXT package. We included 24 mammal species for the analysis that were captured during the entire survey period after excluding captures of cattle (*Bos taurus*) and three species with less than 5 captures (*Didelphis albiventris*, *Coendou prehensilis*, and *Lontra longicaudis)*. The capture data was pooled in 14-day sampling periods to improve homogeneity and to avoid over-dispersion due to short periods of high-activity of a given species (Feng et al., 2021), resulting in 1,485 sampling occasions.

Prior distributions were defined for occupancy and detection intercept estimates (i.e., community mean and detection estimates), represented as normal distributions centred around 0 with a standard deviation of 0.05. We included standardized values of forest cover and forest fragmentation as random effects on species occupancy and detection probability and included location as a fixed effect on community occupancy. In addition, we included the sampling effort as a continuous observation-level covariate on the detection probability to correct for the effect of potential camera malfunctioning. Initial candidate models also included the rate of cattle visits per session and a binary variable to indicate the season, neither of which showed significant effects on occupancy or detection probability, or improved model fit. Species-specific parameters for occupancy and detection were drawn from the community parameter distributions defined in the priors. The model was fitted by looping through each species and session (i.e., combination of a camera trap station and a measurement period of deforestation variables) to estimate the occupancy and detection probabilities for each combination. The model was specified using the logit-link function $logit\,(\varphi) = \log\left(\frac{\varphi}{1-\varphi}\right)$, where φ is the occupancy probability, and the inverse logit-link function $\varphi = \frac{1}{1+\exp\,(-logit(\varphi))}$, which back-transformed the logit-transformed values to ensure that the predicted probabilities are always between 0 and 1. The model was run using three chains of Markov Chain Monte Carlo sampling techniques with 5,000 iterations (2,500 burn-in) to estimate the posterior distributions of the parameters and to obtain estimates of occupancy and detection probabilities for each species-station combination.

We retrieved the community mean of the occupancy probability intercept φ and the occupancy probability intercept per species, the mean community effect of each continuous covariate on the occupancy probability ($\beta_{1,2}$) and its effect on the occupancy probability per species ($\beta_{sp.}$), the effects of the categorical covariate on the community mean of the occupancy probability ($\beta_{station}$), and the fixed effect of the continuous observation-level covariate 'effort' on the detection probability α. The model fit was assessed using Bayesian p-values expressed as the Pearson's χ^2 discrepancy with values outside the range of $0.05 - 0.95$ indicating a lack of fit (Schaub & Kéry, 2012). Effects were considered significant if their 95% Bayesian confidence intervals did not overlap with zero. Our model had an acceptable goodness-of-fit for the entire community (Bayesian p value $= 0.92$) and only failed to generate reliable estimates for the lowland paca *Cuniculus paca* and the white-lipped peccary *Tayassu pecari* (Bayesian p value > 0.95).

2.5.2 COMMUNITY INTEGRITY

We calculated the proportion of forest specialists and disturbance adapted species (i.e., community integrity) in the community for every session. Additionally, we used a general linear model (GLM) to test the relationship between the proportion of forest specialist and disturbance adapted species in response to forest cover and forest fragmentation. The data was checked on assumption associated with linear regression prior to modelling using the protocol of Zuur et al. (2010). We square-transformed data on forest cover to improve linearity and log-transformed data on forest fragmentation to improve normality. Both variables were included in the respective candidate model for forest specialist and disturbance adapted species:

$$\text{Proportion } _{habitat\ affinity} \sim \beta_0 + \beta_1 (\text{forest cover})^2 + \beta_2 \log(\text{forest fragmentation}) + \varepsilon$$

Non-significant covariates were excluded stepwise if their exclusion improved the AIC. The best fitting model was assessed on normality of the residuals, zero inflation, and dispersion using the DHARMa package. The final models were used to predict the effect of deforestation on the proportion of forest specialist and disturbance adapted species. Final models had an acceptable fit (AIC $= 13.69$, Residual deviance $= 3.761$).

3 RESULTS

3.1 COMMUNITY COMPOSITION

We detected a total of 27 native terrestrial mammal species throughout the entire study period (Table 1). 18 species were classified as forest specialists and 9 species as disturbance adapted. Species-specific capture rates were highly heterogenous, ranging from 1 to 623 (mean = 86.4, SD = 154.8) independent events. The most frequently detected species (> 100) were the Collared Peccary *Dicotyles tajacu*, Central American Agouti *Dasyprocta punctata,* Grey Brocket Deer *Mazama gouazoubira,* Jaguar *Panthera onca*, and Lowland Paca *Cuniculus paca*. The Neotropical Otter *Coendou prehensilis,* Brazilian Porcupine *Lontra longicaudis,* and White-eared Opossum *Didelphis albiventris* were the least frequently detected species (< 5). After 1,063 sampling days the entirety of 27 species were detected. On average, stations recorded 14.7 (SD = 8.5) different species and a mean of 242.6 individuals (SD = 295.4). Rarefaction curves show that we likely detected all species present in this community (see appendix III).

Table 1: Species detected in this study, their number of captures, predicted effect of forest cover and forest fragmentation on occupancy, and categorization of habitat affinity in forest specialists (FS) and disturbance adapted species (DA) (n = 27 species). Species are sorted by the approximate adult mean body mass (low to high) .

Species	Trivial name	# captures	Forest cover effect β_1	Fragmentation effect β_2	Habitat affinity
Sciurus ignitus	Bolivian Squirrel	53	**0.55**	0.04	FS
Sylvilagus brasiliensis	Tapeti	14	**0.53**	0.26	FS
Didelphis marsupialis	Common Opossum	5	0.39	0.17	DA
Didelphis albiventris	White-eared opossum	1	/	/	DA
Dasyprocta punctata	Central American Agouti	528	0.11	0.07	FS
Sapajus apella	Tufted Capuchin	17	**0.7**	0.04	FS
Coendou prehensilis	Brazilian porcupine	2	/	/	DA
Euphractus sexcinctus	Yellow Armadillo	7	0.33	0.34	DA
Eira barbara	Tayra	9	**0.49**	0.1	FS
Nasua nasua	South American coati	51	**0.44**	0.04	FS
Procyon cancrivorus	Crab-eating Raccoon	23	0.32	0.29	DA
Leopardus wiedii	Margay	20	**0.56**	0.17	FS
Tamandua tetradactyla	Tamandua	7	0.24	**0.43**	FS

Dasypus novemcinctus	Nine-banded Armadillo	91	0.33	**0.34**	DA
Herpaliurus yagouaroundi	Jaguarundi	15	**0.47**	0.2	FS
Cerdocyon thous	Crab-eating Fox	78	0.28	0.28	DA
Cuniculus paca	Lowland Paca	101	0.14	0.17	FS
Lontra longicaudis	Neotropical otter	1	/	/	FS
Leopardus pardalis	Ocelot	48	**0.46**	0.15	FS
Mazama gouazoubira	Gray Brocket Deer	180	0.07	**0.27**	DA
Dicotyles tajacu	Collared Peccary	623	**0.33**	0.08	FS
Myrmecophaga tridaytyla	Giant Anteater	5	**0.52**	0.14	FS
Tayassu pecari	White-lipped Peccary	49	0.52	0.07	FS
Hydrochoerus hydrochaeris	Capybara	10	0.41	0.32	DA
Puma concolor	Puma	8	**0.45**	**0.01**	FS
Panthera onca	Jaguar	108	**1.08**	0.13	FS
Tapirus terrestris	South American Tapir	43	0.16	0.04	FS

Bold numbers indicate a significant effect on species-specific occupancy, grey numbers with low opacity indicate a too poor model fit. β values are standardized.

3.2 DEFORESTATION

We estimated a mean forest cover of 76.1% (SD = 25.29, ranging from 23.56 to 100) in the buffer zones (0.79 km^2) around camera trap stations in 2017, and a mean of 31.57% (SD = 28.17, ranging from 1.63 to 99.56) in 2021 (see appendix II). Mean forest fragmentation inside buffer zones included an edge length of 9,715 meters (SD = 4086.41, ranging from 4,037 to 15,406) in 2017, and a mean of 11,269 meters (SD = 4086.41, ranging from 2,640 to 20,719) in 2021. On average, all stations lost a mean of 44.5% in forest cover (SD = 22.62) and gained a mean of 1553.38 meters (SD = 7352.86) in edge lengths around forest areas between 2017 and 2021. Forest cover and forest fragmentation were moderately negatively correlated (Pearson's r = -0.46, p < 0.01). The highest forest fragmentation was measured at forest cover values at around 50% and decreased at extreme values of forest cover (see appendix IV).

3.3 COMMUNITY AND SPECIES-SPECIFIC RESPONSES TO DEFORESTATION

The model predicted 27 ± 0.8 species for the entire research area (95% posterior interval) which fits the empirical species richness found in our data and extrapolated rarefaction curves (see appendix III). We found a significantly positive effect of forest cover on occupancy at the community level (β_1 = 0.41 ± 0.19; estimate ± SD) using 110 sessions (Fig. 2a).
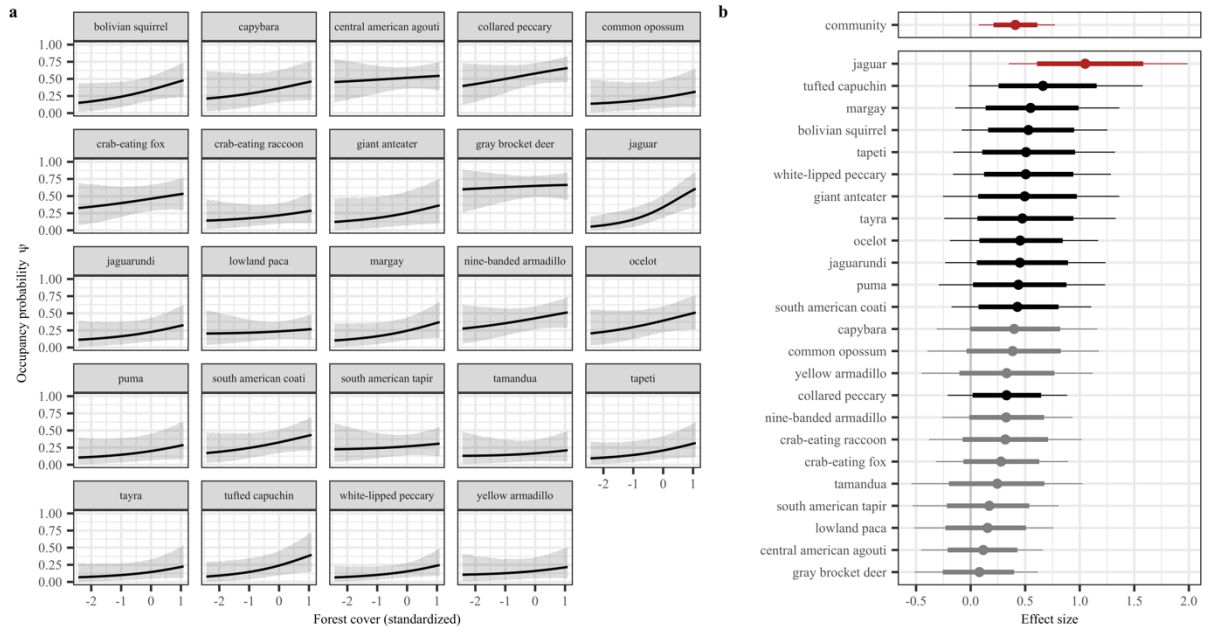
Figure 2 a: Species specific (marginal) effects of forest cover (standardized) on occupancy probability and their standard errors. b: Effect sizes of forest cover on community occupancy probability and species specific occupancy probabilities. Dots represent the mean, bold lines represent the upper bound of the 95% Bayesian confidence interval, thin lines the lower bound. Non-significant effects are coloured grey, significant effects in black, strongly significant effects in red.
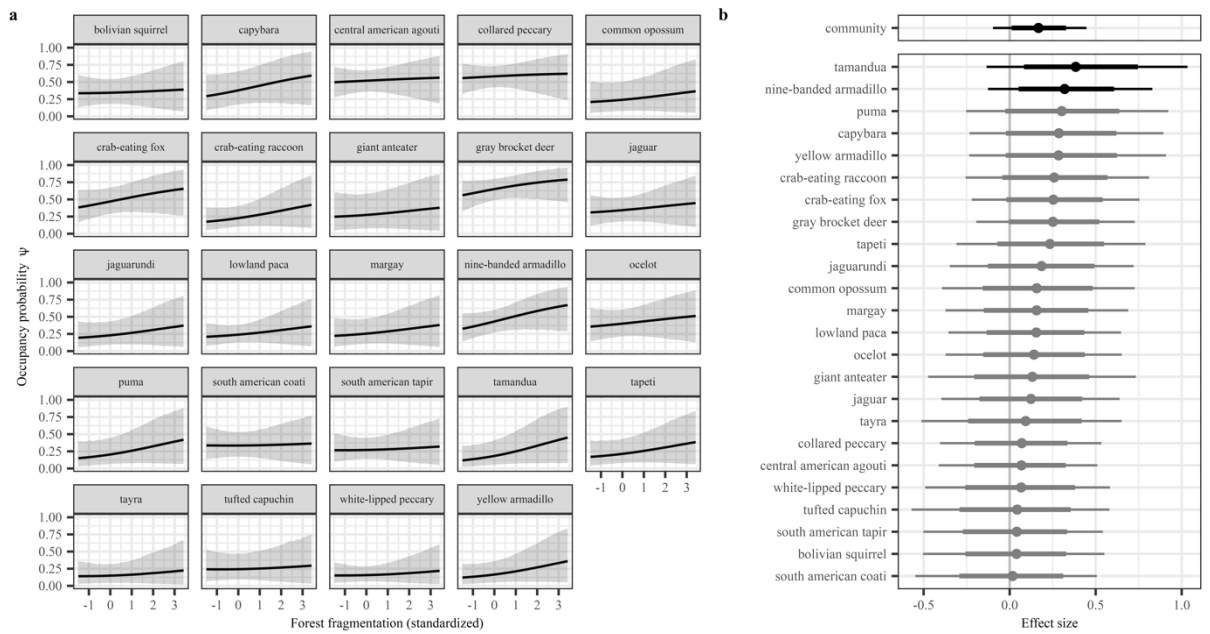
Figure 3 a: Species specific (marginal) effects of forest fragmentation (standardized) on occupancy probability and their standard error. b: Effect sizes of forest fragmentation on community occupancy probability and species specific occupancy probabilities. Dots represent the mean, bold lines represent the upper bound of the 95% Bayesian confidence interval, thin lines the lower bound. Non-significant effects are coloured grey, significant effects in black.

Forest fragmentation too showed a significantly positive but less strong effect on community occupancy ($\beta_2 = 0.18 \pm 0.15$) (Fig. 3b). The session's locations showed unanimous fixed effects

on community occupancy as referenced to G-01. The mean occupancy rate for the community was estimated at $\psi = 0.29 \pm 0.34$. Species-specific occupancy rates varied greatly between 0.13 (Tayra *Eira barbara*) and 0.65 (Grey Brocket Deer *Mazama gouazoubira*) (SD = 0.15). Of the 24 species included in our occupancy model, 13 species were strongly related with forest cover (i.e., 95% Bayesian confidence intervals did not overlap zero), all of which responded positively to increased forest cover (Fig. 2a). No species occupancy showed a negative response to forest cover. The Jaguar *Panthera onca* was found to be most sensitive to forest cover ($\beta_1 = 1.08$). When forest fragmentation increased, occupancy rates at the community level increased as well. However, only the occupancy of Tamandua and Nine-banded armadillo were found to be significantly affected by fragmentation (Fig. 3b) (see also full model parameter estimates in appendix V).

Figure 4: Predicted effect of forest cover on community integrity, i.e., the proportion of forest specialist species (green) and disturbance adapted species (black) present in the community.

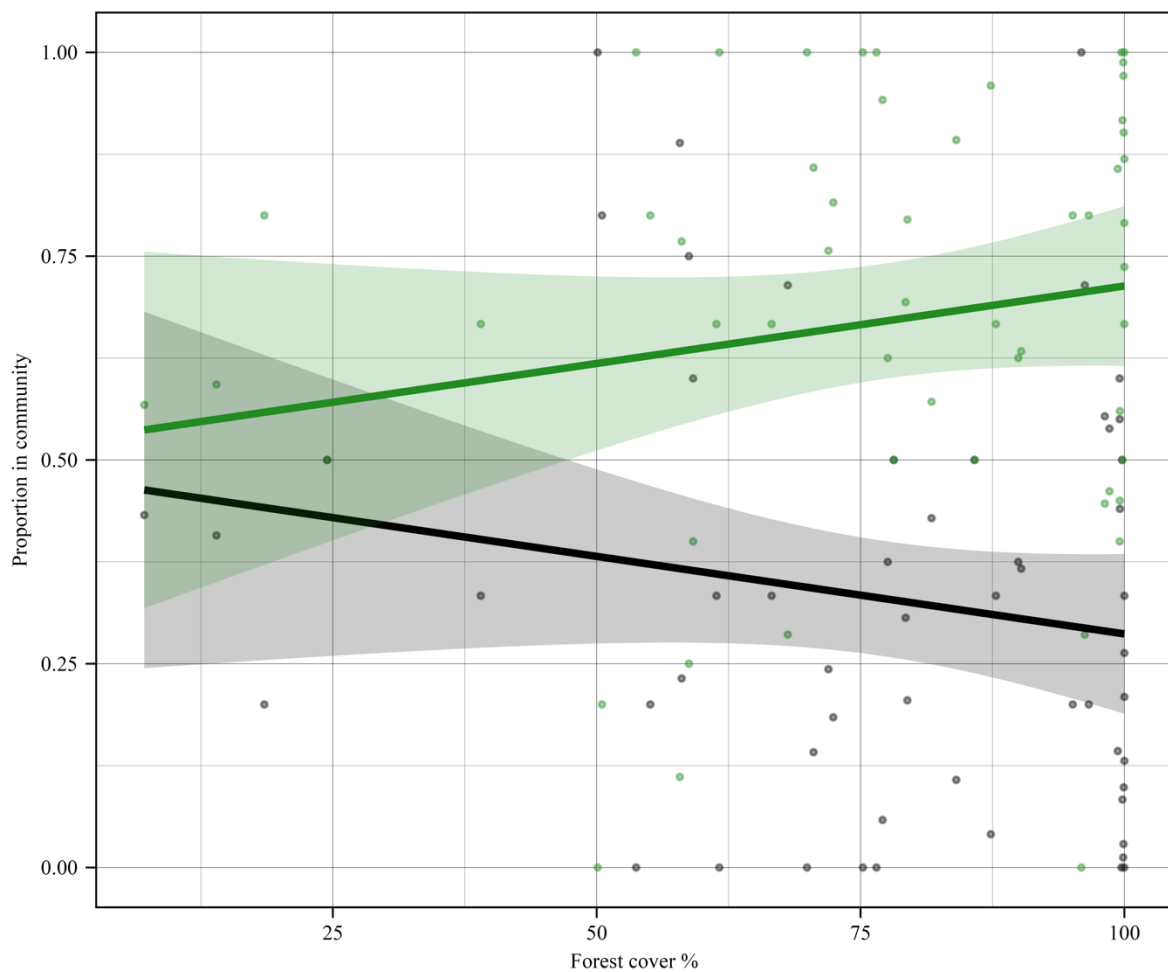The proportion of forest specialist (or disturbance adapted species) did not show a significant response to either changes in forest cover or forest fragmentation, and the exclusion of forest fragmentation improved model fit. Final models on the proportion of forest specialist species only explained a small proportion of the variance in the proportion of functional groups of habitat affinity in the community (3%). Overall, fragmentation was irrelevant in explaining the variance in community integrity and forest cover did not affect the proportion of forest specialists (or disturbance adapted species) in the community significantly (β = 0.0019, SE = 0.0015, p = 0.203) (Fig. 4).

4 DISCUSSION

We analysed the relationship between the effects of deforestation (i.e., forest cover and fragmentation), mammal occupancies, and community integrity (i.e., ratio between specialists and generalists). We investigated a mammal communities at one location over time as site-specific characteristics affect community assembly and the way communities are affected in time (Feng et al., 2021; Kinnaird et al., 2003). For example, the secondary effects of deforestation, like disturbance by human activity often affect mammal communities at a different pace.

As hypothesized, occupancy rates of the majority (76.5%) of forest specialist responded negatively to a reduction in forest cover, as well as the overall occupancy at the community level. This is in line with most studies on the effect of forest cover on community occupancy in the Neotropics (e.g., Amiot et al., 2021; Estavillo et al., 2013) as well as in other tropical forests (e.g., Ramesh et al., 2016; Sosibo et al., 2022; Wearn et al., 2019). One reason for the strong response of forest specialists to a change in forest cover may be the result of trophic cascades (Attayde & Ripa, 2008; Winnie & Creel, 2017) Among forest specialists, forest cover foremostly affects large carnivores that are typically the first species lost at local extinction events (Datta et al., 2008; Michalski & Peres, 2005), which was also evident in our study through high responses in occupancy in carnivore species (e.g., felines). This is because large carnivores have larger homes ranges and territories (Clauss et al., 2010; Ripple et al., 2014) compared to most herbivores (de la Torre et al., 2017; Gavashelishvili & Lukarevskiy, 2008). Habitat availability, however, decreases with a reduction in forest cover (Baldo et al., 2023) and may consequently affect the occupancy of large carnivores. A loss of large carnivores may in turn cause trophic downgrading in ecosystem functioning (Estes et al., 2011). The predation risk (i.e., fear) that mesocarnivores perceive of large carnivores, for example, affected their space use and sequentially mesocarnivore-prey interactions (Suraci et al., 2016). Tropic

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cascades can thus have strong deteriorating effects if altered (Ripple et al., 2014). Compared to other neotropical forests, the CDF harboured a high density of jaguars in 2019 (Meißner et al., 2023). However, with ongoing reductions in forest cover, occupancies of Jaguars and other feline predators (which all responded strongly to forest cover) will likely continue to decline and consequently promote further biodiversity loss.

In contrast to our hypothesis, our results did not suggest a negative effect of forest fragmentation on forest specialist occupancies. Fragmentation seemed to increase occupancy at the community level but only promoted two species-specific occupancies (the Tamandua *Tamandua tetradactyla*, a forests specialist, and the Nine-banded Armadillo *Dasypus novemcinctus,* a disturbance-adapted species). Nine-banded armadillos are known to strive in transition zones between forest zones and grasslands (Feijó et al., 2020; Gallo-Reynoso et al., 2018) and out findings may affirm this species to be edge-tolerant. Tamanduas, however, are not considered edge affine in undisturbed forest (Desbiez & Medri, 2010) and more research is needed to understand how their occupancy responds in fragmented forests. Generally, the effect of fragmentation was weak but no species showed negative responses to increased levels of fragmentation as we had hypothesized. Even though we did not find a negative effect of fragmentation in the CDF, it does affect mammal occupancies negatively at other sites (Andrén, 1994; Bovendorp et al., 2019; Broadbent et al., 2008; Kosydar et al., 2014; Martensen et al., 2012). The effect of fragmentation may be more context depended than that of forest cover because edge habitats that result from fragmentation may also provide novel resources for mammal species, for example through changes in conditions and resources. Thereby, edge habitats may provide novel niches, for example by altering species interactions (e.g., reducing the suitability for other species that exert predative or competitive pressure) (Broadbent et al., 2008; Schneider et al., 2015). Use of edges in highly degraded habitats was for example shown to promote the occupancy of Lemur species while occupancies of closely related sympatrics that did not use edges decreased significantly (Lehman et al., 2006). Occupancies and abundances of large-bodied mammals (e.g., large feline carnivore and ungulate species) are generally reported to decrease with higher levels of fragmentation but were unaffected in our study. Edge habitats were reported to promote the expansion of smaller, more edge-tolerant species including rodents and smaller mesopredators (Broadbent et al., 2008; Schneider et al., 2015), which our results also did not suggest. Therefore, the effects of a reduction in forest cover and increased fragmentation on mammal occupancies seem to be mutually exclusive with forest cover being the stronger determinant. Most fragmentation primarily occurred in areas with an extensive forest coverage (>50%) and may therefore play a subordinated role for of

forest specialists as long as sufficient forest cover is maintained. Species that showed a strong response to a decline in forest cover showed no response to an increase in forest fragmentation and vice versa.

In contrast to forest cover, forest fragmentation did not explain changes in community integrity. However, we also did not find support for our second hypothesis that forest cover significantly affects shifts in the ratio of forest specialist or disturbance adapted species. The observed change in the ratio between specialists and disturbance adapted species corroborated Banks-Leite et al. (2014), even though, our results did not show a shift in this ratio near the threshold of 30% forest cover (Banks-Leite et al., 2014). The proportion of forest specialist species was higher than that of forest generalist in intact forest areas (ca. 75% forest cover) but approached 50% at 25-50% forest cover. Besides, changes in forest cover and forest fragmentation could trigger secondary changes, for example, in environmental characteristics (i.e., microclimate) and human disturbance (e.g., traffic, agriculture) (Easter et al., 2019; Feng et al., 2021; Harihar et al., 2014; Koerner et al., 2017). Studies on the relative importance between forest cover, fragmentation, environment, and human disturbance on mammal occupancies often report the latter to explain the greatest proportion of variance in occupancy patterns, although a large proportion (>50%) frequently remains unexplained (Feng et al., 2021; Ricklefs, 1987; Wearn et al., 2019). The low amount of explained variance in the proportion of forest specialist and disturbance adapted species (3%) is within the expected margin of variance proportions found when forest cover and fragmentation is quantified using remote sensing (e.g., 6% by Feng et al. (2021)). Field-based data on environmental characteristics and human disturbance tend to yield better explanatory powers and can also improve estimates on the effect of vegetation on community assembly compared to proxies like the NDVI (Ahrends et al., 2021). However, the collection of field data on forest cover and fragmentation may be costly, time-consuming, and often difficult to collect if regional authorities or land owners do not cooperate (Chambers et al., 2007).

Furthermore, changes in forest cover and fragmentation are typically the first visible effects of deforestation, while shifts in microclimatic conditions and especially increased human disturbance (i.e., secondary effects) likely occur with a time lag (Metzger et al., 2009). As the time between the primary effects of deforestation and monitoring increases, it becomes increasingly difficult to disentangle primary and secondary drivers of change in community assembly. The strong effect of forest cover reported in literature (e.g., Banks-Leite et al., 2014; Laméris et al., 2020; Martensen et al., 2012; Wallgren et al., 2009; Wearn et al., 2019; Whitworth et al., 2019) raises the question as to whether this effect is actually due to primary

or secondary effects. Study designs in which communities in one location are monitored over time are therefore especially important for separating the primary and secondary effects of changes in mammal communities. Possibly, deforestation at the vegetation level has a lesser effect on mammal communities than secondary influences through disturbance and climatic changes may have. Logged neotropical forests can potentially exceed old-growth forests in productivity and species diversity, but the bioenergetic pathways and ecosystem functioning collapses once agricultural activities started (Malhi et al., 2022). Accordingly, we anticipate that the mammal community in CDF will eventually show the same shift towards disturbance-adapted species as reported by Banks-Leite et al. (2014) after secondary effects continue to take place. We assume the reason that we did not yet detect significant effects on the community integrity therefore to be caused by a temporal lag rather than due to tropical dry forests being more resilient than Atlantic forests. Given the ongoing and extensive deforestation activities in the CDF, its inhabiting mammal communities, and overall ecosystem health face an uncertain future. Our results stress the importance of imminent conservation efforts to preserve the mammalian diversity in the CDF, especially since we demonstrated that community integrity may be maintained even with significant losses in the primary vegetation.

DATA AVAILABILITY STATEMENT

All data and R code used to carry out this study can be accessed via https://github.com/MerlinWe/WildLive

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APPENDIX I: SENTINEL-2 METADATA

Supplementary table 1: Metadata of Sentinel-2 imagery data used in this study.(A) in Period indicates the first half of the year (Jan-Jun), (B) in period indicates the second half of the year (Jul-Dec).

Period	Tile	Image ID	Mission	Resolution	Day of recording
2017 (A)	20LPH	S2A_MSIL1C_20170602T142041_N0205_R010_T20LPH_20170602T142039.SAFE	2A	10	02.06.17 14:20
2017 (A)	20LNH	S2A_MSIL1C_20170602T142041_N0205_R010_T20LNH_20170602T142039.SAFE	2A	10	02.06.17 14:20
2017 (A)	20KNG	S2A_MSIL1C_20170602T142041_N0205_R010_T20KNG_20170602T142039.SAFE	2A	10	02.06.17 14:20
2017 (A)	20KPG	S2A_MSIL1C_20170602T142041_N0205_R010_T20KPG_20170602T142039.SAFE	2A	10	02.06.17 14:20
2017 (B)	20KPG	S2B_MSIL1C_20171015T142029_N0205_R010_T20KPG_20171015T142023.SAFE	2B	10	15.10.17 14:20
2017 (B)	20LNH	S2B_MSIL1C_20171015T142029_N0205_R010_T20LNH_20171015T142023.SAFE	2B	10	15.10.17 14:20
2017 (B)	20LPH	S2B_MSIL1C_20171015T142029_N0205_R010_T20LPH_20171015T142023.SAFE	2B	10	15.10.17 14:20
2017 (B)	20KNG	S2B_MSIL1C_20171015T142029_N0205_R010_T20KNG_20171015T142023.SAFE	2B	10	15.10.17 14:20
2018 (A)	20KPG	S2B_MSIL1C_20180523T142039_N0206_R010_T20KPG_20180523T185544.SAFE	2B	10	23.05.18 14:20
2018 (A)	20LNH	S2B_MSIL1C_20180523T142039_N0206_R010_T20LNH_20180523T185544.SAFE	2B	10	23.05.18 14:20
2018 (A)	20LPH	S2B_MSIL1C_20180523T142039_N0206_R010_T20LPH_20180523T185544.SAFE	2B	10	23.05.18 14:20
2018 (A)	20KNG	S2B_MSIL1C_20180523T142039_N0206_R010_T20KNG_20180523T185544.SAFE	2B	10	23.05.18 14:20
2018 (B)	20KNG	S2A_MSIL1C_20181204T142031_N0207_R010_T20KNG_20181204T160243.SAFE	2A	10	04.12.18 14:20
2018 (B)	20KPG	S2A_MSIL1C_20181204T142031_N0207_R010_T20KPG_20181204T160243.SAFE	2A	10	04.12.18 14:20
2018 (B)	20LPH	S2A_MSIL1C_20181204T142031_N0207_R010_T20LPH_20181204T160243.SAFE	2A	10	04.12.18 14:20
2018 (B)	20LNH	S2A_MSIL1C_20181204T142031_N0207_R010_T20LNH_20181204T160243.SAFE	2A	10	04.12.18 14:20
2019 (A)	20LPH	S2A_MSIL2A_20190612T142041_N0212_R010_T20LPH_20190612T163304.SAFE	2A	10	12.06.19 14:20
2019 (A)	20LNH	S2A_MSIL2A_20190612T142041_N0212_R010_T20LNH_20190612T163304.SAFE	2A	10	12.06.19 14:20

2019 (A)	20KPG	S2A_MSIL2A_20190612T142041_N0212_R010_T20KPG_20190612T163304.SAFE	2A	10	12.06.19 14:20
2019 (A)	20KNG	S2A_MSIL2A_20190612T142041_N0212_R010_T20KNG_20190612T163304.SAFE	2A	10	12.06.19 14:20
2019 (B)	20KPG	S2A_MSIL2A_20190811T142041_N0213_R010_T20KPG_20190811T163606.SAFE	2A	10	11.08.19 14:20
2019 (B)	20LPH	S2A_MSIL2A_20190811T142041_N0213_R010_T20LPH_20190811T163606.SAFE	2A	10	11.08.19 14:20
2019 (B)	20KNG	S2A_MSIL2A_20190811T142041_N0213_R010_T20KNG_20190811T163606.SAFE	2A	10	11.08.19 14:20
2019 (B)	20LNH	S2A_MSIL2A_20190811T142041_N0213_R010_T20LNH_20190811T163606.SAFE	2A	10	11.08.19 14:20
2020 (A)	20LNH	S2A_MSIL2A_20200527T141741_N0214_R010_T20LNH_20200527T182427.SAFE	2A	10	27.05.20 14:17
2020 (A)	20KPG	S2A_MSIL2A_20200527T141741_N0214_R010_T20KPG_20200527T182427.SAFE	2A	10	27.05.20 14:17
2020 (A)	20LPH	S2A_MSIL2A_20200527T141741_N0214_R010_T20LPH_20200527T182427.SAFE	2A	10	27.05.20 14:17
2020 (A)	20KNG	S2A_MSIL2A_20200527T141741_N0214_R010_T20KNG_20200527T182427.SAFE	2A	10	27.05.20 14:17
2020 (B)	20LPH	S2B_MSIL2A_20201009T141739_N0214_R010_T20LPH_20201009T164331.SAFE	2B	10	09.10.20 14:17
2020 (B)	20LNH	S2B_MSIL2A_20201009T141739_N0214_R010_T20LNH_20201009T164331.SAFE	2B	10	09.10.20 14:17
2020 (B)	20KNG	S2B_MSIL2A_20201009T141739_N0214_R010_T20KNG_20201009T164331.SAFE	2B	10	09.10.20 14:17
2020 (B)	20KPG	S2B_MSIL2A_20201009T141739_N0214_R010_T20KPG_20201009T164331.SAFE	2B	10	09.10.20 14:17
2021 (A)	20KNG	S2B_MSIL2A_20210427T141729_N0300_R010_T20KNG_20210427T164150.SAFE	2B	10	27.04.21 14:17
2021 (A)	20LNH	S2B_MSIL2A_20210427T141729_N0300_R010_T20LNH_20210427T164150.SAFE	2B	10	27.04.21 14:17
2021 (A)	20KPG	S2B_MSIL2A_20210427T141729_N0300_R010_T20KPG_20210427T164150.SAFE	2B	10	27.04.21 14:17
2021 (A)	20LPH	S2B_MSIL2A_20210427T141729_N0300_R010_T20LPH_20210427T164150.SAFE	2B	10	27.04.21 14:17
2021 (B)	20LNH	S2B_MSIL2A_20210815T141739_N0301_R010_T20LNH_20210815T164429.SAFE	2B	10	15.08.21 14:17
2021 (B)	20KPG	S2B_MSIL2A_20210815T141739_N0301_R010_T20KPG_20210815T164429.SAFE	2B	10	15.08.21 14:17
2021 (B)	20KNG	S2B_MSIL2A_20210815T141739_N0301_R010_T20KNG_20210815T164429.SAFE	2B	10	15.08.21 14:17
2021 (B)	20LPH	S2B_MSIL2A_20210815T141739_N0301_R010_T20LPH_20210815T164429.SAFE	2B	10	15.08.21 14:17

Cut-off values for NDVI values:

For the wet season, bare soil was defined as < 0.2, sparse vegetation as 0.2 – 0.4, moderate vegetation as 0.4 – 0.6, and dense vegetation as > 0.6. During the dry season, the cut-off values were set as bare soil < 0.2, sparse vegetation 0.2 – 0.3, moderate vegetation 0.3 – 0.5, and dense vegetation > 0.5.

APPENDIX II: STATION-SPECIFIC DEFORESTATION EXTENT
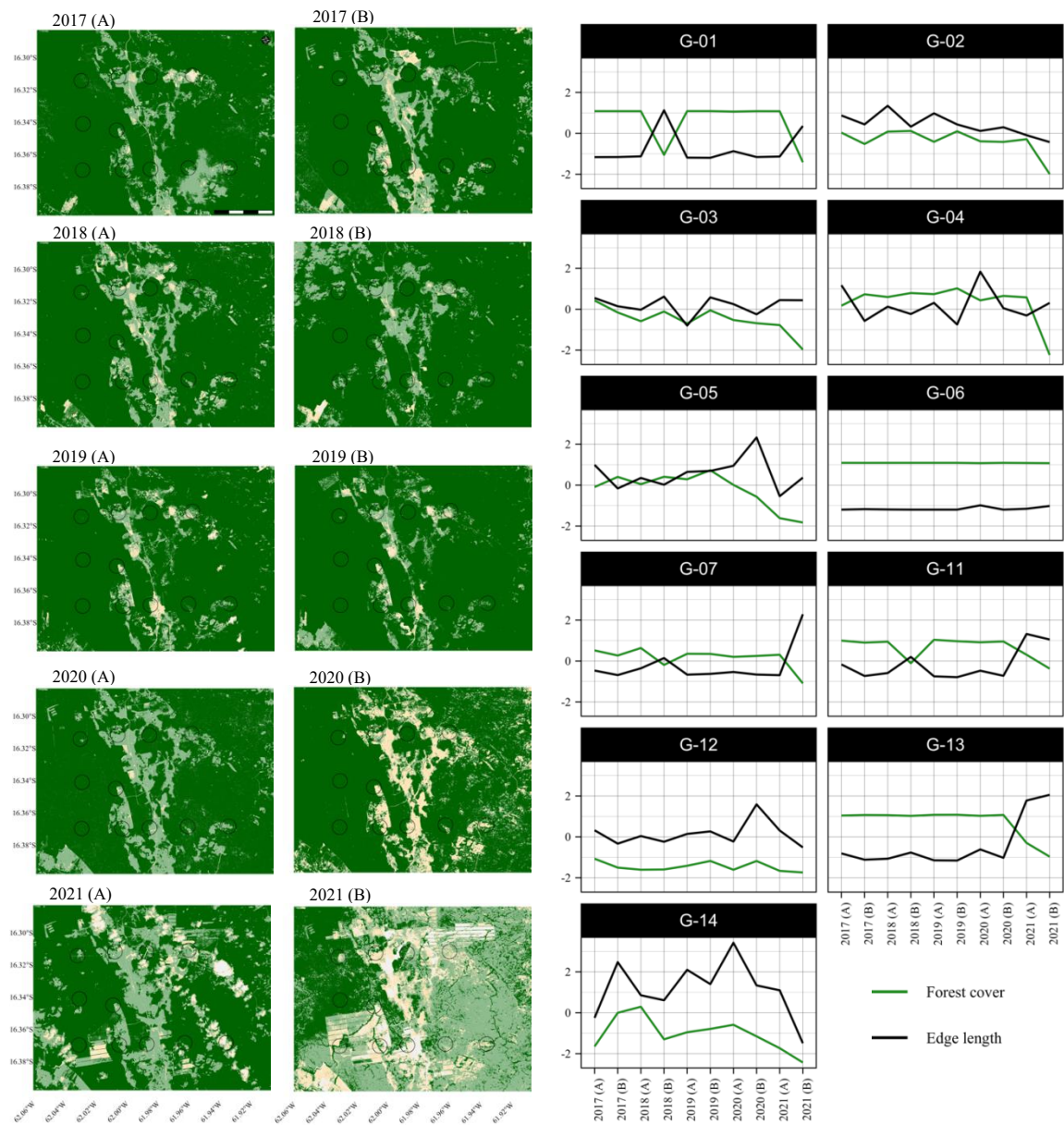
Supplementary figure 1: Deforestation extent per survey period

APPENDIX III: RAREFACTION CURVE OF STUDY COMMUNITY
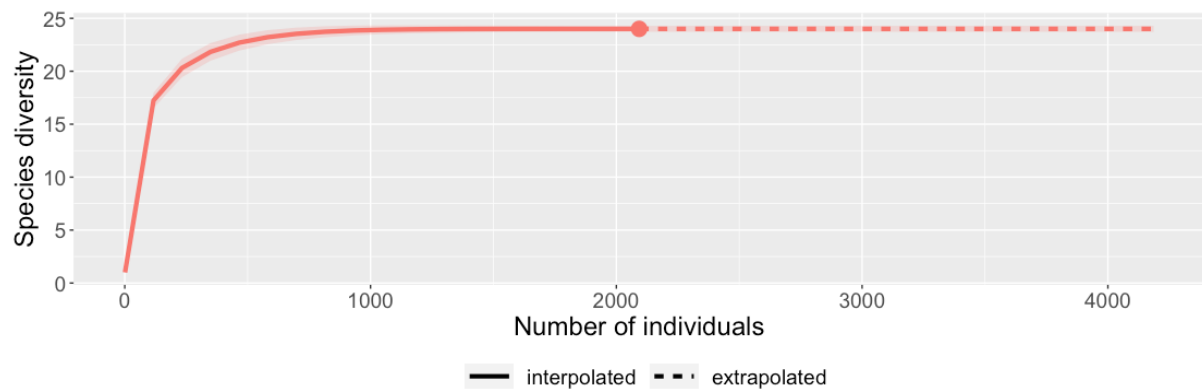
Supplementary figure 2: Extrapolated rarefaction curve of study area

APPENDIX IV: RELATIONSHIP BETWEEN DEFORESTATION VARIABLES

Supplementary figure 3: Scatter plot of deforestation covariates along with a loess fit line.

APPENDIX V: OCCUPANCY MODEL PARAMETER ESTIMATES

Supplementary table 2: Parameter estimates of the multi-species occupancy model used in this study.

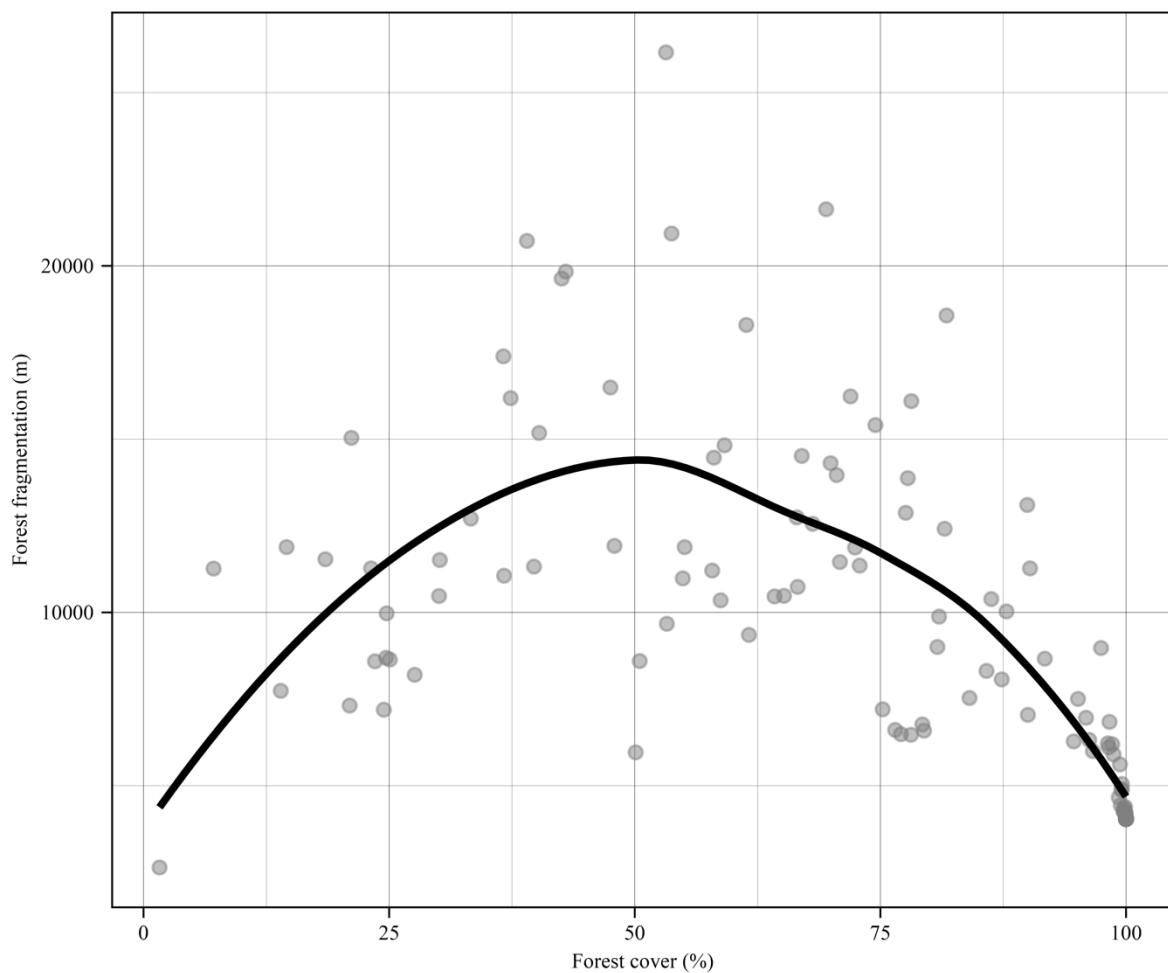|	Mean	SD	Naive SE	Time-series SE
Bayesian p value (Community)	0.92	0.14	0.01	0.00
Bayesian p value (Central American agouti)	0.89	0.22	0.02	0.00
Bayesian p value (South american coati)	0.64	0.48	0.01	0.01
Bayesian p value (Tamandua)	0.63	0.48	0.01	0.01
Bayesian p value (Capybara)	0.10	0.30	0.00	0.00
Bayesian p value (Bolivian squirrel)	0.94	0.24	0.00	0.00
Bayesian p value (Collared peccary)	0.73	0.41	0.02	0.00
Bayesian p value (Crab-eating fox)	0.53	0.50	0.01	0.01
Bayesian p value (Lowland paca)	0.99	0.10	0.00	0.00
Bayesian p value (Gray brocket deer)	1.00	0.05	0.00	0.00
Bayesian p value (Ocelot)	0.74	0.44	0.01	0.01
Bayesian p value (Jaguar)	0.84	0.04	0.00	0.00
Bayesian p value (Puma)	0.66	0.47	0.01	0.01
Bayesian p value (Nine-banded armadillo)	0.85	0.36	0.00	0.00
Bayesian p value (Jaguarundi)	0.66	0.48	0.01	0.01
Bayesian p value (Common opossum)	0.49	0.50	0.01	0.01
Bayesian p value (South american tapir)	0.89	0.32	0.00	0.00
Bayesian p value (Tayra)	0.77	0.42	0.01	0.01
Bayesian p value (Margay)	0.57	0.50	0.01	0.01
Bayesian p value (Crab-eating racoon)	0.51	0.50	0.01	0.01
Bayesian p value (White-lipped peccary)	0.97	0.18	0.00	0.00
Bayesian p value (Tufted capuchin)	0.55	0.50	0.01	0.01
Bayesian p value (Yellow armadillo)	0.65	0.48	0.01	0.01
Bayesian p value (Giant anteater)	0.58	0.49	0.01	0.01
Bayesian p value (Tapeti)	0.64	0.48	0.01	0.01
Occupancy probability (Central american agouti)	0.52	0.33	0.00	0.01
Occupancy probability (South american coati)	0.33	0.42	0.01	0.02

Occupancy probability (Tamandua)	0.16	0.60	0.01	0.02
Occupancy probability (Capybara)	0.36	0.55	0.01	0.02
Occupancy probability (Bolivian squirrel)	0.34	0.42	0.01	0.01
Occupancy probability (Collared peccary)	0.58	0.34	0.00	0.01
Occupancy probability (Crab-eating fox)	0.47	0.40	0.01	0.02
Occupancy probability (Lowland paca)	0.23	0.38	0.00	0.01
Occupancy probability (Gray brocket deer)	0.65	0.36	0.00	0.01
Occupancy probability (Ocelot)	0.39	0.43	0.01	0.01
Occupancy probability (Jaguar)	0.33	0.39	0.00	0.02
Occupancy probability (Puma)	0.18	0.63	0.01	0.02
Occupancy probability (Nine-banded armadillo)	0.43	0.36	0.00	0.01
Occupancy probability (Jaguarundi)	0.22	0.52	0.01	0.02
Occupancy probability (Common opossum)	0.22	0.62	0.01	0.02
Occupancy probability (South american tapir)	0.27	0.41	0.01	0.01
Occupancy probability (Tayra)	0.13	0.60	0.01	0.02
Occupancy probability (Margay)	0.24	0.47	0.01	0.02
Occupancy probability (Crab-eating racoon)	0.21	0.46	0.01	0.01
Occupancy probability (White-lipped peccary)	0.15	0.43	0.01	0.01
Occupancy probability (Tufted capuchin)	0.23	0.53	0.01	0.02
Occupancy probability (Yellow armadillo)	0.15	0.60	0.01	0.02
Occupancy probability intercept (Giant anteater)	0.23	0.72	0.01	0.04
Occupancy probability intercept (Tapeti)	0.15	0.53	0.01	0.02
Fixed effect of location G-01 on occupancy probability	0.00	0.00	0.00	0.00
Fixed effect of location G-02 on occupancy probability	0.84	0.32	0.00	0.02
Fixed effect of location G-03 on occupancy probability	-2.51	0.48	0.01	0.02
Fixed effect of location G-04 on occupancy probability	-1.36	0.33	0.00	0.02
Fixed effect of location G-05 on occupancy probability	-1.61	0.37	0.00	0.02
Fixed effect of location G-06 on occupancy probability	-0.62	0.29	0.00	0.01
Fixed effect of location G-07 on occupancy probability	0.61	0.29	0.00	0.01
Fixed effect of location G-11 on occupancy probability	-0.86	0.29	0.00	0.01
Fixed effect of location G-12 on occupancy probability	-6.79	2.17	0.03	0.05

Fixed effect of location G-13 on occupancy probability	-2.04	0.36	0.00	0.01
Fixed effect of location G-14 on occupancy probability	-7.13	2.16	0.03	0.05
Community effect of forest cover on occupancy probability	0.41	0.18	0.00	0.01
Forest cover effect on occupancy probability (Central american agouti)	0.11	0.28	0.00	0.01
Forest cover effect on occupancy probability (South american coati)	0.44	0.32	0.00	0.01
Forest cover effect on occupancy probability (Tamandua)	0.24	0.40	0.01	0.01
Forest cover effect on occupancy probability (Capybara)	0.41	0.37	0.00	0.01
Forest cover effect on occupancy probability (Bolivian squirrel)	0.55	0.35	0.00	0.01
Forest cover effect on occupancy probability (Collared peccary)	0.33	0.28	0.00	0.01
Forest cover effect on occupancy probability (Crab-eating fox)	0.28	0.31	0.00	0.01
Forest cover effect on occupancy probability (Lowland paca)	0.14	0.32	0.00	0.01
Forest cover effect on occupancy probability (Gray brocket deer)	0.07	0.29	0.00	0.01
Forest cover effect on occupancy probability (Ocelot)	0.46	0.34	0.00	0.01
Forest cover effect on occupancy probability (Jaguar)	1.08	0.42	0.01	0.02
Forest cover effect on occupancy probability (Puma)	0.45	0.38	0.00	0.01
Forest cover effect on occupancy probability (Nine-banded armadillo)	0.33	0.31	0.00	0.01
Forest cover effect on occupancy probability (Jaguarundi)	0.47	0.37	0.00	0.01
Forest cover effect on occupancy probability (Common opossum)	0.39	0.39	0.01	0.01
Forest cover effect on occupancy probability (South american tapir)	0.16	0.34	0.00	0.01
Forest cover effect on occupancy probability (Tayra)	0.49	0.40	0.01	0.01
Forest cover effect on occupancy probability (Margay)	0.56	0.38	0.00	0.01
Forest cover effect on occupancy probability (Crab-eating racoon)	0.32	0.35	0.00	0.01
Forest cover effect on occupancy probability (White-lipped peccary)	0.52	0.37	0.00	0.01
Forest cover effect on occupancy probability (Tufted capuchin)	0.70	0.40	0.01	0.01
Forest cover effect on occupancy probability (Yellow armadillo)	0.33	0.39	0.01	0.01
Forest cover effect on occupancy probability (Giant anteater)	0.52	0.41	0.01	0.01
Forest cover effect on occupancy probability (Tapeti)	0.53	0.38	0.00	0.01
Community effect of edge length on occupancy probability	0.19	0.15	0.00	0.01
Edge length effect on occupancy probability (Central american agouti)	0.07	0.24	0.00	0.01
Edge length effect on occupancy probability (South american coati)	0.01	0.28	0.00	0.01
Edge length effect on occupancy probability (Tamandua)	0.43	0.31	0.00	0.01

Edge length effect on occupancy probability (Capybara)	0.32	0.29	0.00	0.01
Edge length effect on occupancy probability (Bolivian squirrel)	0.04	0.27	0.00	0.01
Edge length effect on occupancy probability (Collared peccary)	0.08	0.24	0.00	0.01
Edge length effect on occupancy probability (Crab-eating fox)	0.28	0.26	0.00	0.01
Edge length effect on occupancy probability (Lowland paca)	0.17	0.27	0.00	0.01
Edge length effect on occupancy probability (Gray brocket deer)	0.27	0.24	0.00	0.01
Edge length effect on occupancy probability (Ocelot)	0.15	0.26	0.00	0.01
Edge length effect on occupancy probability (Jaguar)	0.13	0.26	0.00	0.01
Edge length effect on occupancy probability (Puma)	0.34	0.30	0.00	0.01
Edge length effect on occupancy probability (Nine-banded armadillo)	0.34	0.25	0.00	0.01
Edge length effect on occupancy probability (Jaguarundi)	0.20	0.29	0.00	0.01
Edge length effect on occupancy probability (Common opossum)	0.17	0.30	0.00	0.01
Edge length effect on occupancy probability (South american tapir)	0.04	0.28	0.00	0.01
Edge length effect on occupancy probability (Tayra)	0.10	0.30	0.00	0.01
Edge length effect on occupancy probability (Margay)	0.17	0.28	0.00	0.01
Edge length effect on occupancy probability (Crab-eating racoon)	0.29	0.28	0.00	0.01
Edge length effect on occupancy probability (White-lipped peccary)	0.07	0.28	0.00	0.01
Edge length effect on occupancy probability (Tufted capuchin)	0.04	0.30	0.00	0.01
Edge length effect on occupancy probability (Yellow armadillo)	0.32	0.30	0.00	0.01
Edge length effect on occupancy probability (Giant anteater)	0.14	0.31	0.00	0.01
Edge length effect on occupancy probability (Tapeti)	0.26	0.28	0.00	0.01

APPENDIX VI: SOFTWARE PACKAGES

Supplementary table 3: R packages used for data mining, data preparation, and data analysis used in this study and their references.

Package	Reference
camtrapR	Jürgen Niedballa, Rahel Sollmann, Alexandre Courtiol, Andreas Wilting (2016). camtrapR: an R package for efficient camera trap data management. Methods in Ecology and Evolution 7(12), 1457-1462.
camtrapRdeluxe	Juergen Niedballa, Alexandre Courtiol and Rahel Sollmann (2018). camtrapRdeluxe: A R package that builds on camtrapR to provide more functionalities and flexibility. R package version 0.99.10.9999.
cowplot	Claus O. Wilke (2020). cowplot: Streamlined Plot Theme and Plot Annotations for 'ggplot2'. R package version 1.1.1. https://CRAN.R-project.org/package=cowplot

data.table | Matt Dowle and Arun Srinivasan (2021). data.table: Extension of `data.frame`. R package version 1.14.2. https://CRAN.R-project.org/package=data.table

DHARMa | Florian Hartig (2022). DHARMa: Residual Diagnostics for Hierarchical (Multi-Level / Mixed) Regression Models. R package version 0.4.5. https://CRAN.R-project.org/package=DHARMa

exifr | Dewey Dunnington and Phil Harvey (2021). exifr: EXIF Image Data in R. R package version 0.3.2. https://CRAN.R-project.org/package=exifr

gdata | Gregory R. Warnes, Ben Bolker, Gregor Gorjanc, Gabor Grothendieck, Ales Korosec, Thomas Lumley, Don MacQueen, Arni Magnusson, Jim Rogers and others (2017). gdata: Various R Programming Tools for Data Manipulation. R package version 2.18.0. https://CRAN.R-project.org/package=gdata

geosphere | Robert J. Hijmans (2021). geosphere: Spherical Trigonometry. R package version 1.5-14. https://CRAN.R-project.org/package=geosphere

ggeffects | Lüdecke D (2018). "ggeffects: Tidy Data Frames of Marginal Effects from Regression Models." Journal of Open Source Software_, *3*(26), 772. doi: 10.21105/joss.00772 (URL https://doi.org/10.21105/joss.00772).

ggpubr | Alboukadel Kassambara (2020). ggpubr: 'ggplot2' Based Publication Ready Plots. R package version 0.4.0. https://CRAN.R-project.org/package=ggpubr

ggrepel | Kamil Slowikowski (2021). ggrepel: Automatically Position Non-Overlapping Text Labels with 'ggplot2'. R package version 0.9.1. https://CRAN.R-project.org/package=ggrepel

ggsn | Oswaldo Santos Baquero (2019). ggsn: North Symbols and Scale Bars for Maps Created with 'ggplot2' or 'ggmap'. R package version 0.5.3. https://github.com/oswaldosantos/ggsn

ggspatial | Dewey Dunnington (2021). ggspatial: Spatial Data Framework for ggplot2. R package version 1.1.5. https://CRAN.R-project.org/package=ggspatial

GISTools | Chris Brunsdon and Hongyan Chen (2014). GISTools: Some further GIS capabilities for R. R package version 0.7-4. https://CRAN.R-project.org/package=GISTools

gridExtra | Baptiste Auguie (2017). gridExtra: Miscellaneous Functions for "Grid" Graphics. R package version 2.3. https://CRAN.R-project.org/package=gridExtra

Hmisc | Frank E Harrell Jr (2021). Hmisc: Harrell Miscellaneous. R package version 4.6-0. https://CRAN.R-project.org/package=Hmisc

iNEXT | T. C. Hsieh, K. H. Ma and Anne Chao. 2020 iNEXT: iNterpolation and EXTrapolation for species diversity. R package version 2.0.20 URL: http://chao.stat.nthu.edu.tw/wordpress/software-download/.

jsonlite | Jeroen Ooms (2014). The jsonlite Package: A Practical and Consistent Mapping Between JSON Data and R Objects. arXiv:1403.2805 [stat.CO] URL https://arxiv.org/abs/1403.2805

knitr | Yihui Xie (2021). knitr: A General-Purpose Package for Dynamic Report Generation in R. R package version 1.36.

lubridate | Garrett Grolemund, Hadley Wickham (2011). Dates and Times Made Easy with lubridate. Journal of Statistical Software, 40(3), 1-25. URL https://www.jstatsoft.org/v40/i03/

plyr | Hadley Wickham (2011). The Split-Apply-Combine Strategy for Data Analysis. Journal of Statistical Software, 40(1), 1-29. URL http://www.jstatsoft.org/v40/i01/

purr | Lionel Henry and Hadley Wickham (2020). purrr: Functional Programming Tools. R package version 0.3.4. https://CRAN.R-project.org/package=purrr

raster | Robert J. Hijmans (2021). raster: Geographic Data Analysis and Modeling. R package version 3.5-9. https://CRAN.R-project.org/package=raster

reshape2 | Hadley Wickham (2007). Reshaping Data with the reshape Package. Journal of Statistical Software, 21(12), 1-20. URL http://www.jstatsoft.org/v21/i12/.

rgdal	Roger Bivand, Tim Keitt and Barry Rowlingson (2021). rgdal: Bindings for the 'Geospatial' Data Abstraction Library. R package version 1.5-27. https://CRAN.Rproject.org/package=rgdal
rgeos	Roger Bivand and Colin Rundel (2021). rgeos: Interface to Geometry Engine - Open Source ('GEOS'). R package version 0.5-8. https://CRAN.R-project.org/package=rgeos
rnaturalearth	Andy South (2017). rnaturalearth: World Map Data from Natural Earth. R package version 0.1.0. https://CRAN.R-project.org/package=rnaturalearth
rnaturalearthdata	Andy South (2017). rnaturalearthdata: World Vector Map Data from Natural Earth Used in 'rnaturalearth'. R package version 0.1.0. https://CRAN.R-project.org/package=rnaturalearthdata
scales	Hadley Wickham and Dana Seidel (2022). scales: Scale Functions for Visualization. R package version 1.2.0. https://CRAN.R-project.org/package=scales
sen2r	L. Ranghetti, M. Boschetti, F. Nutini, L. Busetto (2020). sen2r: An R toolbox for automatically downloading and preprocessing Sentinel-2 satellite data. Computers & Geosciences, 139, 104473. doi: 10.1016/j.cageo.2020.104473, URL: https://sen2r.ranghetti.info/.
sf	Pebesma, E., 2018. Simple Features for R: Standardized Support for Spatial Vector Data. The R Journal 10 (1), 439-446, https://doi.org/10.32614/RJ-2018-009
sp	Pebesma, E.J., R.S. Bivand, 2005. Classes and methods for spatial data in R. R News 5 (2), https://cran.r-project.org/doc/Rnews/.
stringr	Hadley Wickham (2019). stringr: Simple, Consistent Wrappers for Common String Operations. R package version 1.4.0. https://CRAN.R-project.org/package=stringr
tidyr	Hadley Wickham (2021). tidyr: Tidy Messy Data. R package version 1.1.4. https://CRAN.R-project.org/package=tidyr
tidyverse	Wickham et al., (2019). Welcome to the tidyverse. Journal of Open Source Software, 4(43), 1686, https://doi.org/10.21105/joss.01686
zoo	Achim Zeileis and Gabor Grothendieck (2005). zoo: S3 Infrastructure for Regular and Irregular Time Series. Journal of Statistical Software, 14(6), 1-27. doi:10.18637/jss.v014.i06