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An evaluation of the efficiency of survey methods to monitor large mammals in Cusuco National Park, Honduras

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Preface

I have desired to be part of conservation team undertaking research in a rainforest environment. Joining a project undertaken in Cusuco National Park in Honduras provided me with that opportunity. The field work was undertaken in cooperation with Operation Wallacea a non-governmental conservation research organisation who has carried out surveys in Cusuco National Park in Honduras annually since 2004.

I would like to especially thank my supervisors Torbjørn Haugaasen (NMBU) and Richard Bischof (NMBU). I also want to thank the Opwall team Neil Reid, Danielle Gilroy, Hanna Hoskins, Sam Puls and Eric McNeill. Thanks to Richard Bischof for help with the statistical analysis and for developing the statistical model. Thanks to Torbjørn Haugaasen for the help with the valuable comments and feedback during the writing process. Thank you to the Opwall team for providing the opportunity for me to carry out field work with them in their field season in 2016. Thank you to Sam Puls for all the valuable help in the field and for teaching me how to use QGIS.

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Abstract

Field sign surveys and camera trap surveys are frequently used in the tropics as the preferred survey technique to monitor large terrestrial mammal communities. However, these methods vary in the efficiency of detecting certain species. Investigating these variations are important to improve survey efforts, survey design and efficiency of detecting focal species. Cusuco National Park is part of the Mesoamerican hotspot, inhabiting a large mammal community important for ecosystem functions. The study mainly aimed to investigate variation in species detection probabilities according to survey method. The study also aimed to get an understanding of the large mammal composition in the park. Finally, this study aimed to examine the temporal variation in species detection probabilities for field sign surveys. Field sign surveys and camera trap surveys have been undertaken annually in the park since 2014. Field sign surveys proved as the most effective method for surveying Baird's tapir, Central American agouti, nine-banded armadillo, white-nosed coati and grey fox. For spotted paca, small felid sp. and tayra, camera trap surveys proved to be the most effective method. None of the methods proved more effective than the other for surveying deer sp. and collared peccary. Baird's tapir was found to be locally declining in Cusuco. Combining methods to survey large mammal species maximized monitoring efficiency in Cusuco National Park.

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1. Introduction

Large mammals affect tropical forest ecosystems through several ecological functions. Predators regulate prey populations and may indirectly affect prey behaviour (Ray et al. 2013), while herbivores affect the vegetation community through grazing, seed predation and seed dispersal (Haugaasen et al. 2010; Kuprewicz 2013; Paine & Beck 2007). Additionally, large mammals may cause soil disturbance affecting nutrient cycling (Kuprewicz 2013). Large forest dwelling mammals are declining throughout their range as a consequence of increased human population growth, habitat destruction, global climate change, and overexploitation, which affect both animal populations and ecosystems (Primack 2012). In comparison to smaller mammals, large mammals are especially predisposed to declines and sensitive to offtakes because of their life history traits, such as large body size, slow reproduction and large home ranges (Purvis 2001; Wilkie et al. 2011). The significant roles of large mammals in forest ecosystems and the consequences of declining populations, highlight the importance of monitoring large mammal communities to aid conservation management (Primack 2012; Tobler et al. 2008; Voss & Emmons 1996).

Camera traps and field sign have frequently been used to survey large mammals in the tropics (Tobler et al. 2008; Trolle 2003; Voss & Emmons 1996). Searching for field signs is one of the oldest methods to detect mammal presence in an area (Gopalaswamy et al. 2012; Voss & Emmons 1996). Recording field signs, such as dung and tracks, is therefore often used as a relatively cheap and efficient survey approach, but relies on suitable field conditions and trained field personnel (Ahumada et al. 2013; Burnham et al. 1980; Gopalaswamy et al. 2012). Due to these challenges, camera traps are often implemented as a survey technique, where field personnel error can be reduced to correct placement, maintenance of cameras and identification of photographs (Ahumada et al. 2013). Recent advances in camera trapping technology have made cameras cheaper and even more sensitive to detect multiple species of medium- to large-bodied mammals, as well as birds, and in some instances even arboreal mammals (O'Connell et al. 2011; Rovero et al. 2010). Camera trapping is non-invasive, there is no observer bias, it is time and cost effective, easy to standardize and easy to replicate in time and space (Rovero et al. 2010; Tobler et al. 2008). It is possible to monitor nocturnal, diurnal and rare animals, while it provides a record of time, date, location, species and in some cases individual recognition (Ahumada et al. 2011; O’Brien et al. 2011; O’Connell & Bailey 2011; Rovero & Marshall 2009; Rovero et al. 2010; Rovero et al. 2013). Several studies have investigated the variation in success of camera trap and field sign in surveying large mammals (Mazzolli et al. 2016; Munari et al. 2011; Nichols et al. 2008; Silveira et al. 2003). Survey methods do not perform equally well in all situations and for every species in a study system (Munari et al. 2011; Silveira et al. 2003). Factors such as time constraints, environment, costs, study aims and target species of interest is necessary to understand to select the appropriate method (Silveira et al. 2003).

From sign survey- and camera trap -data a variety of measures, such as estimates of mammal density (Keeping 2014), abundance (O’Brien 2011), species richness (Rovero et al. 2014), diversity (O'Brien et al. 2010) and occupancy (Ahumada et al. 2013) can be obtained. An examination of some of these measures require large populations, capture-recapture data and individual recognition of mammals (O’Connell & Bailey 2011; Rovero & Marshall 2009). Occupancy can be a useful alternative in situations where animals cannot be individually identified and when animal densities are low (MacKenzie et al. 2002; Mazzolli & Hammer

2013; O'Connell & Bailey 2011). Since occupancy models only require species occurrence information, it is often cheaper and more efficient compared to abundance and animal density measures (O'Connell & Bailey 2011). With the right study design, occupancy information can provide knowledge on species distribution, habitat preference, population dynamics, species interactions and even abundance (MacKenzie & Nichols 2004; O'Connell & Bailey 2011; Rovero et al. 2014; Thorsen 2016). In an occupancy framework, the ecological process (occupancy probability) and the observation process (detection probability) are modelled together (Kéry & Schaub 2011). Detection probability is often not incorporated in occupancy studies, but is important because both processes together produce the observed data (Kéry & Schaub 2011; MacKenzie et al. 2002). Sampled data is commonly represented by only a few individuals from the true population, because not all individuals will be detected with absolute certainty during data collection (O'Brien et al. 2011; Pollock et al. 2002). It is therefore important that detection probability (p) of a species can be incorporated too account for these imperfect detections and avoid biased estimates (MacKenzie et al. 2006). Detection probability can be defined as the probability of detecting a species at a site given that the species is present (O'Brien 2011). Most studies focus on the ecological process of occupancy, but by understanding detection probabilities the issues of false negatives (not detecting species that are present) can possibly be avoided in surveys (Delaney & Leung 2010). Investigating species detection probability could aid the ability to detect rare species and species with naturally low densities which in turn can guide conservation management (Delaney & Leung 2010). Detection probability is an important parameter to estimate when species are detected imperfectly (MacKenzie et al. 2006; O'Connell & Bailey 2011). Non-detection of a species does not necessarily mean absence, which is especially the case when mammal numbers are low and the risk of not detecting a species becomes high.

Central America supports high species richness, as it is part of the Mesoamerican Biological Corridor. The region only represents 0.5 % of the world's land surface, but due to the variation of habitats and the link to the North- and South-American continents, it is home to approximately 7% of the world's biological diversity (Miller et al. 2001). Deforestation of the forests in Mesoamerica is occurring at an alarming rate and approximately 80% of the forest cover may have already lost or fragmented (Martin & Blackburn 2009). A key biodiversity area within this region is Cusuco National Park in Honduras. This park is part of the Mesoamerican biodiversity hotspot, which indicates an area with unique and substantial species richness (Field & Long 2007; The Opwall Trust 2017). To the best of my knowledge there are only one other study undertaken, investigating large mammal communities in Honduras (Gonthier & Castañeda 2013). The limited information available highlights the need to study these communities. The reason for this may be that Honduras is considered as one of the most dangerous countries to travel to. Despite years of collecting data on large mammals in Cusuco National Park, there is no study investigating the large mammal community using both field sign and camera trap surveys. Investigating differences in efficiency of these survey approaches can improve survey efforts, survey strategy and effectiveness of detecting target species (Munari et al. 2011). The main aim of the study was to investigate differences between camera trap and field sign surveys by examining species composition as well as detection probabilities for species and specie- groups. In addition, the study aimed to examine the temporal variation in species detection probabilities for field sign surveys.

2. Methods

2.1 Study area

This study was conducted in Cusuco National Park located in northwest Honduras (15°33'53.0"N 88°18'16.4"W) (Figure 1). The park was established in 1987 and is managed by Corporacion Hondurena de Desarollo Forestal (CODEFOR) together with a few non-governmental organizations, such as Operation Wallacea (Field & Long 2007). The 234.4 km² national park is connected to the Merendon mountain range, which stretches from Guatemala into Honduras (Butler 2013; Green et al. 2012). The park is considered a cloud forest and elevation ranges from 60m in the west to 2425m in the center (Appendix A)(Martin & Blackburn 2009). The establishment structure of Cusuco divided the park into a core zone (76.9 km²) surrounded by a buffer zone (156.5 km²). The core zone was established to ensure preservation of biodiversity, while the buffer zone was developed for sustainable forest management such as shaded coffee plantations and sustainable logging practices (Figure 1) (Martin & Blackburn 2009). The main forest types in the park are semi-arid pine forest, moist pine forest, moist broadleaf forest and dwarf forest (Field & Long 2007; Green et al. 2012). At lower elevation, climate is hot most of the year with occasional showers during the rainy season. At higher elevation, the temperature is mild with cool nights, rain and occasional storms for the majority of the year. Average rainfall recorded during field seasons in 2014-2016 was 5mm per 24 hours. Average temperatures measured during field seasons in 2014-2016 was 18 °C during 24 hours (Wallacea 2014-2016).

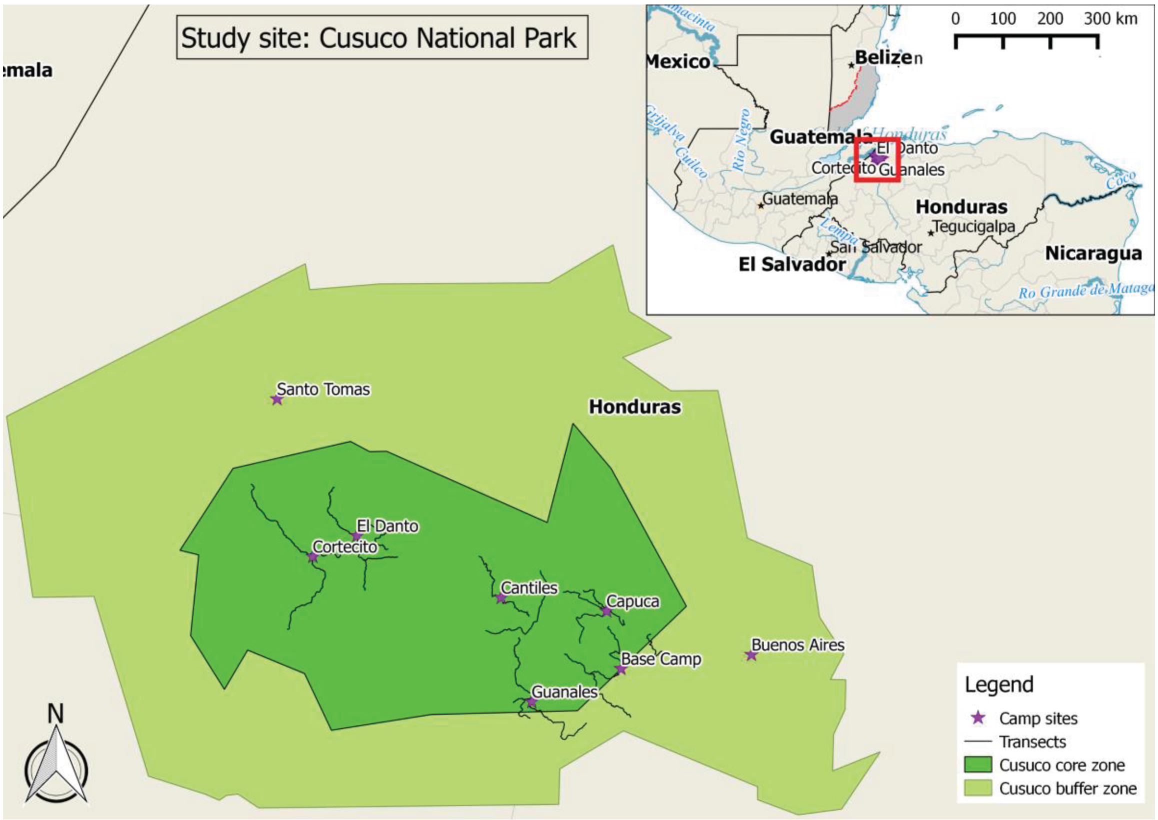

Figure 1: Cusuco National Park is located northwest in Honduras, Central America (15°33'53.0"N 88°18'16.4"W). The map includes camp sites, transects and parks division into a core zone (76.9 km²) and a buffer zone (156.5 km²).

2.2 Data collection

2.2.1 Field sign surveys

Operation Wallacea (Opwall), a non-governmental conservation research organisation, has undertaken monitoring surveys in Cusuco National Park in Honduras since 2004. Mammal surveys have been carried out annually since 2006, utilizing a standardized method recording the presence of mammals using field signs. In 2016, data was collected in collaboration with Opwall from June 15 to August 9. Five camps were set up in the park and each camp had 3-4 already established 3km transects for surveys. A local guide accompanied the surveys and to ensure annual consistency of field sign identifications, the same guide was used every year. For each transect survey, we recorded date, start and end time, camp, transect, weather condition, and number of observers. Transects were walked slowly from start to the end, making direct observations and searching for signs such as vocalizations, dung, hair, leftover food, and tracks of large mammals on the transect (Green et al. 2012; Reid 2016b). Upon detection of a field sign, we registered distance from beginning of transect, species, detection cue and spatial coordinates. Additionally, a photo was taken of each sign with a ruler for reference (Figure 2). Any indicator of human disturbance, such as hunter platforms, snares or encounters with locals were also registered.

Figure 2: Coati diggings from field sign surveys

2.2.2 Camera trap surveys

Camera traps were applied as an additional survey method to field sign surveys to detect large mammals in 2014. In 2014, cameras were placed randomly throughout the park to investigate species detections. In 2015 and 2016, camera trap surveys were designed to investigate the effects of disturbance on transects and to calculate abundance using the Random Encounter Model (REM) (Reid 2016a; Rowcliffe et al. 2008). The main difference between 2014 camera trap design and the design from 2015 and 2016, was that fewer cams were placed in 2014, but they were left in the field for a longer period of time. In 2016, three cameras were installed perpendicular to each transect at distances of <20m, ~150m ~300m, at two different locations along the transect, as designed by Reid (2016b) (Figure 3). Six cameras were installed on each transect. Cameras were placed a minimum of 200m from other transects, and 200m from 2015 camera locations. Off-trail camera placement was restrained by the hilly landscape, and camera sites were selected according to local topography and the guides' expertise.

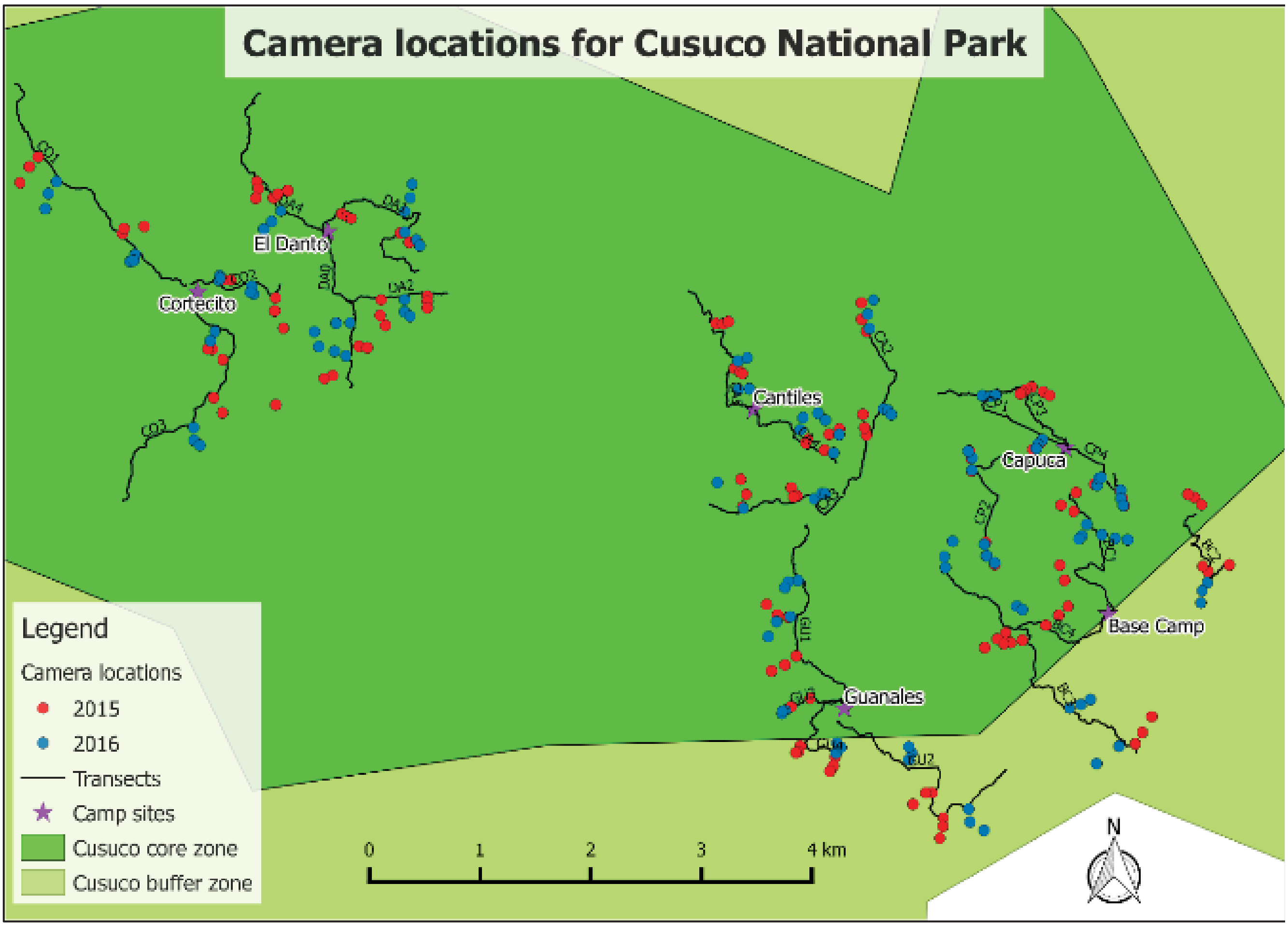

Figure 3: Study area including cams sites, transects and camera locations for 2015 and 2016

Cameras were placed on a tree approximately 1m off the ground using clip-bands attached to the cameras. They were positioned and tilted to cover the largest area possible according to vegetation and incline (Figure 4). Vegetation was cleared 3m in front of the camera to avoid obstruction (Reid 2016b; Rovero et al. 2010). For each camera, spatial coordinates, camera ID, camp, sub-site ID, time and date for setup and retrieval, were recorded (Figure 4) (Rovero et al. 2010). A hand-held GPS unit was used to record camera trap locations (Reid 2016b). Lures and bait are commonly used in camera trapping studies, but to promote random sampling and to avoid species biases, no bait or lures were used in this study (Ahumada et al. 2011; MacKenzie & Royle 2005; Rowcliffe et al. 2008). Cameras were operating for 24 hours, to detect both diurnal and nocturnal species (Rovero et al. 2010).

Figure 4: Positioning camera traps on trees approximately 1m above the ground

Bushnell Trophy CAM HD trail cameras were used at all locations. These are passive infrared digital cameras that detect both movement and changes in heat relative to the surrounding temperatures. They record videos when warm-blooded animals walk past the camera using infrared light. Infrared cameras are often preferred in surveys, because they are cheaper, without flash and easier to use compared to active systems (Mazzolli & Hammer 2013; Rovero et al. 2010). Cameras were set to have a high trigger speed so the camera would start recording at the first sign of movement. High trigger speed is often preferred for faunal surveys to increase the likelihood to detect elusive species (Rovero et al. 2010). A delay time of 1 minute was set between video recordings, which is considered the lowest delay time sufficient for an animal to move on and to avoid recaptures of the same individual (Rovero et al. 2010). Videos were set to record for 20sec and the date and time of detection was printed on each video. Further details on camera settings can be found in Appendix B. Due to constraints regarding available cameras, time and accessibility to remote areas in the park, the 24 cameras available were deployed sequentially throughout the park rather than simultaneously. Such constraints can be common when undertaking camera trapping studies (Ahumada et al. 2011; Rovero et al. 2014). For each camp site, 1-4 transects had 3-6 cameras operating for three days (Figure 3). Most cameras were placed inside the core zone of the park, because of the risk of cameras being stolen in the buffer zone, where there is more human activity. The camera trap arrays cover approximately 77 km². As some cameras were stolen or stopped functioning, there were 22 operating camera sites in 2015, 128 in 2015 and 111 in 2016.

2.3 Statistical analysis

All statistical computation was carried out in R (R Core Team 2017) using R studio, an interface for R (RStudio Team 2016).

2.3.1 Data preparation

Field sign survey data

Species identification from field signs was carried out by the guide and the large mammal research staff *in-situ*. Data from 2014 to 2016 was included in the occupancy analysis. The raw presence only data were converted into detection/non-detection data for sites and for each species. To get these values from the field sign data, each transect were divided into sites of 0-500m along the transect. The transect were again divided into sub-sites of 0-100m. The data was arranged in R to acquire the number of sub-sites a species was detected at within each site (y) along a transect. These counts provide us with the necessary spatial information to include detection probability in the model. A y cell was given a value between 0 and 5 for a specific site depending on the number of sub-sites the species was detected at. The data also included the total number of sub-sites possible for detection (N). The resulting data set included year, site ID, transect ID, species ID, sub-site detection counts and total sub-sites within the site.

Camera trap survey data

For camera trap surveys, species captured by the cameras were identified *in-situ* by the large mammal research-staff and myself. A bird expert from the bird team was consulted to ensure precise identification of bird detections. By combining raw presence only data with camera setup information in R, it was possible to acquire counts of how many days a species was detected (y) at a camera location. For 2014, the count data could have y values between 0-28 and for 2015 and 2016 the detection data could have y values between 0-3. The data also included the total number of days a camera was active. The resulting data included site ID, year, transect species/functional group ID, counts of days detected (y) and the duration a camera was active (N). By arranging the data in this way, analyzing consecutive videos of the same individual as multiple events was avoided, thus making events independent. Obtaining events according to hours is usually enough to avoid analyzing recaptures (Rovero et al. 2014; Tobler et al. 2008). These events provide the necessary temporal information to include detection probability in the model.

2.3.2 Occupancy analysis and modelling

A hierarchical single-species, single-season site occupancy model was used for each method to estimate species detection probabilities (p) (Bischof et al. 2014). Assumptions necessary to estimate occupancy are closure, site independence and lack of false positives. Closure means that the mammal community is not changing within a survey season, and false positives means that a species absence is not mistaken for being present (Kéry & Schaub 2011; MacKenzie et al. 2002; Tobler et al. 2008). In tropical forests, populations are often closed and few are migratory (Rowcliffe et al. 2008). With a survey duration over maximum 60 days and camera activity only for 3 days, the closure assumption is likely met in the current study (Rowcliffe et al. 2008; Tobler et al. 2008). Species difficult to distinguish from each other are grouped to decrease the risk of false positives. Site independence is, on the other hand, not met due to the proximity of survey sites and transects (O'Connell & Bailey 2011). Occupancy estimates retrieved from the analysis can still indicate species spatial use, but to avoid any misinterpretation of this term, this study will focus exclusively on species detection probabilities. Occupancy models can be developed using a frequentist framework investigating Maximum likelihood estimates, or a Bayesian framework investigating posterior distribution estimates (Kéry & Schaub 2011). A Bayesian modelling framework has recently become popular due to the development of computer software that can handle complex Bayesian inference problems (Kéry & Schaub 2011; MacKenzie et al. 2006). Bayesian framework have been found to work better when sample sizes are small and outputs are more explanatory compared to a frequentist framework. It is 95% likely that the correct estimate is found within the confidence interval. A narrow critical interval (CI) indicates higher accuracy of estimates compared to wide critical intervals. A Bayesian framework includes a prior probability for the parameter, which affects the posterior probability distribution of the parameters (MacKenzie 2006).

Simulated data were used in the occupancy model to test model accuracy. The posterior distribution estimates matched the input values from the provided data, giving validity to the model (Kéry & Schaub 2011). The models were run in R studio using the statistical software Just Another Gibbs Sampler (JAGS) (Plummer 2003), running through packages R2jags (Su & Yajima 2015) and rjags (Plummer et al. 2016). The model was designed to acquire species probability estimates of detection (p) and local occupancy/use according to sites (Ψ) and transects $(\emptyset)$. In a hierarchical model, the ecological process, the true occurrence of a species at a site (z_s) or transect (z_t) can be modelled as a Bernoulli process which is affected by the local occupancy probability of a site (Ψ) or transect $(\emptyset)$:

$$z_t \mid \emptyset \sim Bernoulli(\emptyset)$$

$$z_s \mid \Psi, z_t \sim Bernoulli(\Psi * z_t)$$

The observation process is modeled separately as a binomial process where the counts of detection events y is affected by the actual occurrence at a site (z_s) and the detection probability (p) according to the number of total subsites/days duration (N).

$$y \mid p, z_s \sim Binomial(z_s * p, N)$$

The main model was modelled for large mammal species and functional groups without any covariates as a null model, where occupancy and detection probability is assumed to be constant (Rovero et al. 2014). Non-informative priors were used for all models 0,1 for all parameters. Markov Chain Monte Carlo (MCMC) simulations were run with 3 chains of 60 000 iterations each, and a burnin rate of 15 000 (first 15 000 iterations discarded) iterations and with a thinning rate of 5 (every fifth iteration discarded). This returned 27 000 samples from the posterior distribution.

2.3.3 Species

Of the 20 mammal species detected in Cusuco National Park between 2014 and 2016, this study will focus on 13 large terrestrial mammals: Baird's tapir (*Tapirus bairdii*), red brocket deer (*Mazama americana*), white-tailed deer (*Odocoileus virginianus*), spotted paca (*Cuniculus paca*), collared peccary (*Pecari tajacu*) , Central American agouti (*Dasyprocta punctata*), nine-banded armadillo (*Dasypus novemcinctus*), jaguarundi (*Herpailurus yagouaroundi*), ocelot (*Leopardus pardalis*), margay (*Leopardus wiedii*), white-nosed coati (*Nasua narica*), tayra (*Eira barbara*) and grey fox (*Urocyon cinereoargenteus*) (shortened names will be used for simplicity throughout the paper). Examples of species detected can be found in (Figure 5) and the full list of species recorded can be found in Appendix C. White-tailed deer and red brocket deer field signs can be difficult to distinguish from each other, so these species were grouped into their family group deer sp. (Cervidae sp.) for method comparison. The detection history indicates that of these detections are likely to be red brocket deer (Appendix D). Margay, ocelot and jaguarundi was also grouped together into family group small felid sp. (Felidae sp.), because of the issues of identifying species from field signs. These family groups will be compared to other species recorded and will therefore be termed as species throughout the study for simplicity, even though these are family groups. When camera trap surveys are discussed individually species-specific names will be used, since species are easier to identify from videos.

Species were also grouped to account for low detections of certain species: large herbivores (>20kg, tapir, white-tailed deer, red brocket deer and white collared peccary), small herbivores (<20kg, paca and agouti), omnivores (coati, tayra, grey fox and armadillo) and carnivores (ocelot, margay and jaguarundi) following Ahumada et al. (2013). Since there were only small felids considered as carnivores in the study undertaken between 2014 and 2016, carnivores and small felid sp. will include the same species. They will still be discussed I different settings to maintain the possibility for comparison. In the temporal analysis undertaken between 2006 and 2016 puma (*puma concolor*) was detected and is therefore included in the carnivore group. If jaguar (*Panthera onca*) was detected, it would also be included in the carnivore group.

Temporal evaluation was only undertaken for field sign surveys, since data have been collected for ten years, while camera trap surveys have only been undertaken for three years. Species with the highest detections and with a limited number of years where detections were below 5 was included (tapir, paca, armadillo, coati, carnivore, and deer sp.). Data from 2007 and 2013 were not included in analysis because the data was incomplete. For the detection probability analysis, species with ≥ 5 detections during at least one year and for one method was included in the analysis. In years and for methods where those species have fewer detections than 5, detection probability was considered as 0 or data deficient (DD) to simplify the comparison, and to acquire mean detection probabilities. Species with less than 5 detections were not included in the analysis because if they were, detection probabilities would be mostly explained by the priors and less explained by the data (Kéry & Schaub 2011).

Figure 5: Example pictures from camera trap videos; from top left red brocket deer, spotted paca, baird's tapir, margay, collared peccary and white-nosed coati.

3. Results

3.1 Summary

Field sign surveys accrued a total census effort of approximately 126 km between 2014 and 2016, yielding 401 detections. The 261 operating cameras from 2014 to 2016 accumulated 1138 camera trap days (mean = 4.36 active days per camera) and yielding a total of 914 video recordings (mean = 3.5 videos per camera). The surveys combined detected thirteen large mammal species (532 detections), two arboreal species (30 detections), three medium-sized mammals (58 detections), three small mammals (321 detections) and sixteen bird species (331 detections). Of the ten clearly identifiable species deer sp (Cervidae sp.) had the highest number of detections followed by white-nosed coati (Nasua narica), spotted paca (Cuniculus paca) and Baird's tapir (Tapirus bairdii) (Table 1). Of the species groups, large herbivores had the highest number of detections (231 detections) followed by omnivores (161 detections), small herbivores (123 detections) and carnivores (15 detections) (Appendix C).

3.2 Species detection probability

Of the 10 large mammal species, only eight could be included in the occupancy analysis to acquire detection probabilities. Tayra and grey fox were not included in the model due to the low number of detections. Similarly, some species included in the model had too few detections to be analysed in certain years or by a certain method (Table 2). Deer sp. and coati were the only species with enough detections for all years and for both methods. Summarizing average detection probabilities from both methods, paca (0.225) had the highest overall detectability followed by deer sp. (0.216), coati (0.204), armadillo (0.148), tapir (0.114), peccary (0.095), agouti (0.036) and small felids (0.04) (Table 2). The species-group with the highest overall detection probability was omnivores (0.277), followed by small herbivores (0.270), large herbivores (0.260) and carnivores (0.040) (Table 2). Occupancy estimates for site and transects are included in (Appendix E).

Table 1: Species, species group, conservation status, detection summaries for field sign surveys and camera trap surveys from 2014 to 2016 and detection frequency from camera trap surveys

Family and scientific name	***Common name***	***Species group***	***IUCN status***	***Camera trap***	***Field sign***	***Total***	***Camera trap Frequency***
Tapiridae *Tapirus bairdii*	Baird's tapir	Large herbivore	EN	10	54	64	8.79
Cervidae sp.	Deer sp.	Large herbivore	NA	37	84	121	32.51
Mazama americana	Red brocket deer	Large herbivore	DD	36	52	88	31.6
Odocoileus virginianus	White-tailed deer	Large herbivore	LC	1	32	33	1.14
Cuniculidae *Cuniculus paca*	Spotted paca	Small herbivore	LC	74	11	85	65
Tayassuidae *Pecari tajacu*	Collared peccary	Large herbivore	LC	28	18	46	24.6
Dasyproctidae *Dasyprocta punctata*	Central American agouti	Small herbivore	LC	3	37	40	2.63
Dasypodidae *Dasypus novemcinctus*	Nine-banded armadillo	Omnivore	LC	4	49	53	3.51
Felidae sp.	Small felids	Carnivore	NA	13	2	15	11.42
Herpailurus yagouaroundi	Jaguarundi	Carnivore	LC	1	0	1	1.14
Leopardus pardalis	Ocelot	Carnivore	LC	1	1	2	1.14
Leopardus wiedii	Margay	Carnivore	NT	11	1	12	9.66
Procyonidae *Nasua narica*	White-nosed coati	Omnivore	LC	30	72	102	26.36
Mustelidae *Eira barbara*	Tayra	Omnivore	LC	3	0	3	2.63
Canidae *Urocyon cinereoargenteus*	Grey fox	Omnivore	LC	0	3	3	0
Total detections				202	330	532	
Total Species				9	9	10	

EN = endangered; VU = vulnerable; NT = near threatened; DD = data deficient; LC = least concern; NA = not applicable.
Frequency = Number of detections per 1000 trap nights (1.138).

Table 2: Mean detection probabilities (95% critical intervals) from the posterior distribution for field sign and camera trap surveys between 2014-2016. Species and species groups with a minimum of 5 detections for one of the survey methods and for one of three years were included in the analysis.

Mammal Species	**Camera trap surveys p**				**Field sign surveys p**				**Total**
	2014	2015	2016	Average mean	2014	2015	2016	Average mean	
Baird's tapir	0.047 (0.011- 0.115)	DD	DD	0.016	0.163 (0.086- 0.272)	0.130 (0.069- 0.221)	DD	0.098	**0.114**
Deer sp.	0.086 (0.040-0.145)	0.090 (0.018-0.279)	0.105 (0.028-0.267)	0.094	0.112 (0.072- 0.126)	0.173 (0.097- 0.277)	0.080 (0.032- 0.170)	0.122	**0.216**
Spotted paca	0.099 (0.061-0.145)	0.265 (0.117-0.437)	0.229 (0.080-0.437)	0.198	DD	DD	0.080 (0.024- 0.198)	0.027	**0.225**
Collared peccary	0.049 (0.020-0.092)	0.097 (0.019- 0.290)	DD	0.049	DD	0.138 (0.005- 0.516)	DD	0.046	**0.095**
Central-American Agouti	DD	DD	DD	DD	0.108 (0.068- 0.169)	DD	DD	0.036	**0.036**
Nine-banded armadillo	DD	DD	DD	DD	0.107 (0.032- 0.244)	0.094 (0.039- 0.193)	0.243 (0.120- 0.390)	0.148	**0.148**
Small felid sp.	0.034 (0.007- 0.088)	0.086 (0.020- 0.257)	DD	0.04	DD	DD	DD	DD	**0.040**
White-nosed coati	0.049 (0.020-0.091)	0.092 (0.018-0.286)	0.098 (0.018-0.305)	0.080	0.213 (0.129-0.312)	0.062 (0.021-0.149)	0.096 (0.050-0.172)	0.124	**0.204**
Large Herbivore	0.085 (0.049- 0.129)	0.092 (0.029- 0.231)	0.106 (0.027- 0.271)	0.094	0.175 (0.125-0.240)	0.214 (0.152-0.292)	0.108 (0.050-0.208)	0.166	**0.260**
Small herbivore	0.091 (0.057-0.132)	0.265 (0.116-0.440)	0.266 (0.104-0.470)	0.207	0.109 (0.068-0.168)	DD	0.080 (0.024-0.201)	0.063	**0.270**
Omnivore	0.053 (0.028- 0.087)	0.096 (0.017-0.288)	0.099 (0.019-0.307)	0.083	0.250 (0.171- 0.335)	0.128 (0.065-0.224)	0.204 (0.127-0.298)	0.194	**0.277**
Carnivore	0.034 (0.007- 0.088)	0.086 (0.020- 0.257)	DD	0.04	DD	DD	DD	DD	**0.040**

DD = data deficient (<5 detections)

3.3 Comparing sign and camera trap surveys

Number of species detected were the same for both survey methods (9) and the number of species increased by one when combining species. Estimated average detection probabilities for camera trap surveys were considerably higher for paca and small felids compared to field sign surveys. Estimated average detection probability for field sign surveys were reasonably higher for tapir, agouti, armadillo and coati in comparison to camera trap surveys (Figure 6). Estimated average detection probabilities for deer species and peccary were similar between the two survey approaches (Figure 6). For species-groups camera traps average estimated detection probabilities were higher for small herbivores and carnivores, while field sign detection probabilities were higher for omnivores and large herbivores (Figure 7).

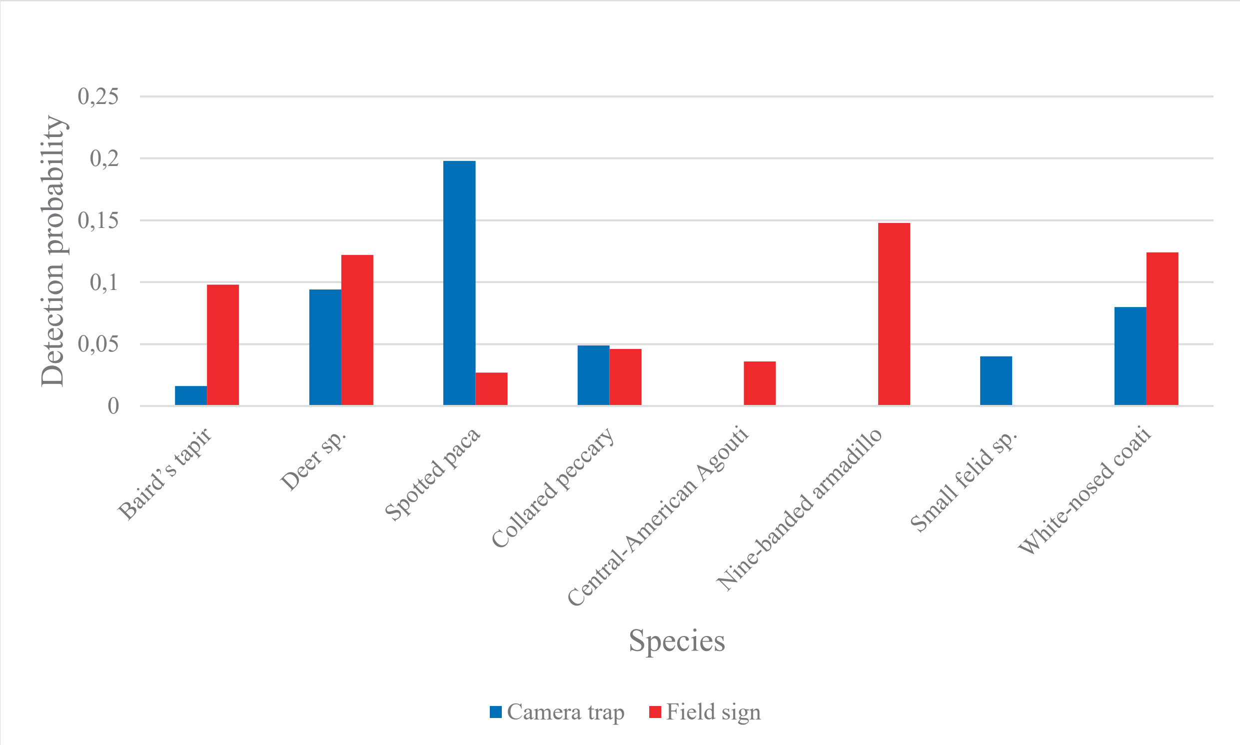

Figure 6: Detection probability for taxa included in the occupancy analysis for camera trap surveys and field sign surveys in Cusuco National Park

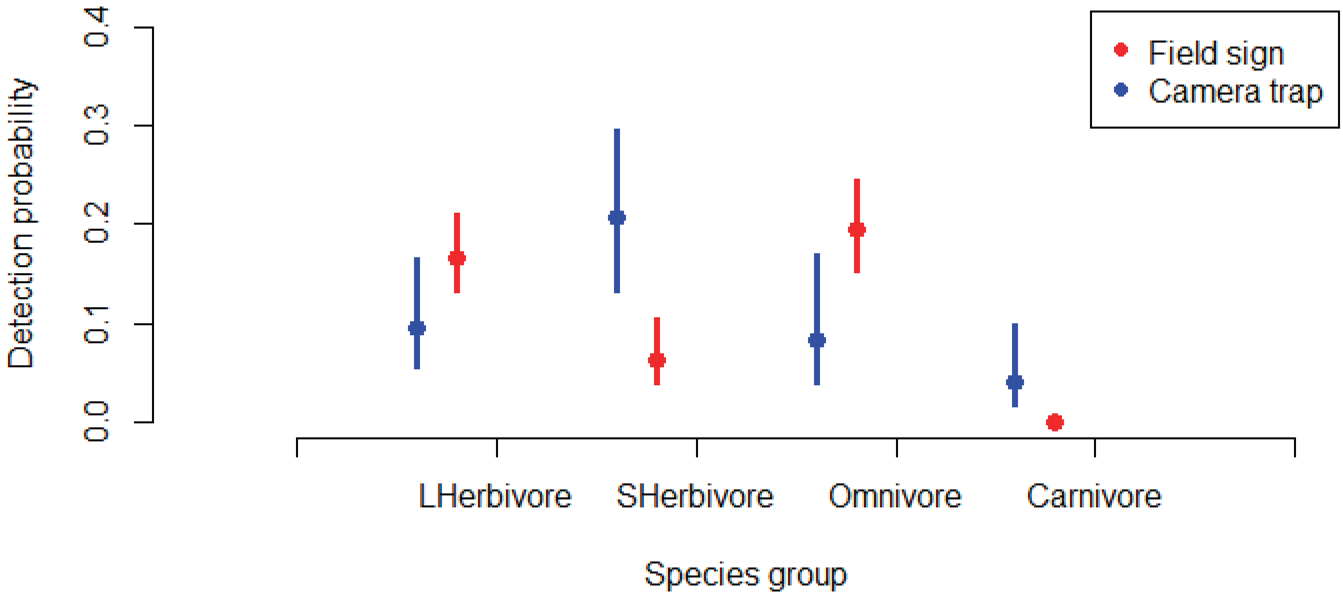

Figure 7: Comparing detection probability between survey methods and functional guild. Values are retrieved from the null model averaged over three years (2014-2016). Detection probability posterior mean from camera trap surveys are shown as blue points with posterior critical intervals as blue lines. Detection probability posterior mean from field sign surveys are shown as red points with posterior critical intervals as red lines.

3.4 Temporal variation in detection probability for field signs

Estimated detection probabilities for tapir show a declining trend since 2006. Small felid sp. estimated detection probabilities appear stable until 2012. However, between 2014-2016 there were not enough detections to analyse felid detection probabilities, indicating a decline in detections after 2012. The low detection probabilities for paca in 2014 and 2015 may not be of great importance, since detections increased again in 2016 and detections from camera traps were high (>15) in 2014 and 2015 (Figure 8, Appendix C). Detection probability fluctuations look relatively stable across years for deer sp., white-nosed coati and armadillo.

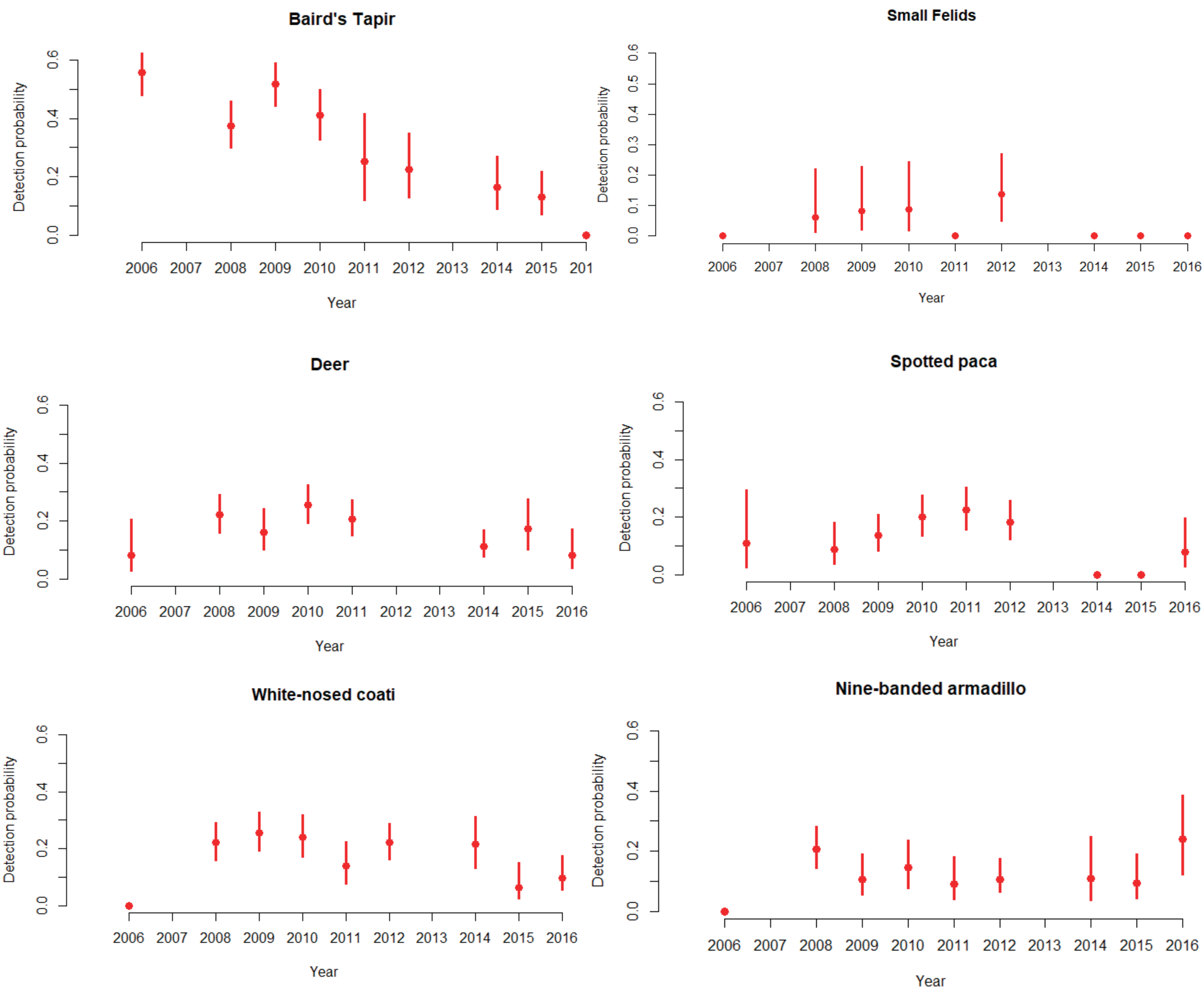

Figure 8: Temporal variation in detection probability for selected species. Detection probability posterior mean is shown as red points with posterior 95% critical interval (CI) as red lines. Single red dots represent detections 0-4. Detections from 2007 and 2013 are not included due to incomplete data for these years

4. Discussion:

4.1 The Cusuco mammal community

Results show that a variety of large mammal species were detected during the three years of surveying in Cusuco National Park. The large mammal community in The Rio Plátano Biosphere Reserve in eastern Honduras also investigated the large mammal community using camera traps (Gonthier & Castañeda 2013). The study recorded higher number of species, different species composition and difference in detection frequencies in comparison to Cusuco (Gonthier & Castañeda 2013). In The Rio Plátano, puma, striped hog-nosed skunk (*Conepatus semistriatus*) and giant anteater (*Myrmecophaga tridactyla*) was recorded, while these were not detected in Cusuco (Gonthier & Castañeda 2013). In Cusuco, jaguarundi and margay were recorded, but they were not detected in The Rio Plátano (Gonthier & Castañeda 2013). The Rio Plátano had higher frequency of species such as tapir, white-tailed deer, agouti and ocelot, while Cusuco had higher frequencies of red brocket deer, paca, peccary, armadillo, white-nosed coati and tayra (Gonthier & Castañeda 2013) (Table 1). For some of these species the differences in frequencies were quite high and it would be interesting to investigate these variations further. For example, why are agouties one of the most frequently recorded species in other studies, but very few were recorded by camera traps in Cusuco (Gonthier & Castañeda 2013; Munari et al. 2011; Srbek-Araujo & Chiarello 2005)?. Why are margay detected regularly in Cusuco compared to ocelot being the most frequently detected small cat in other studies (Gonthier & Castañeda 2013; Munari et al. 2011; Srbek-Araujo & Chiarello 2005)?. These differences might be linked to habitat (Rovero et al. 2014), elevation (Ahumada et al. 2013), human disturbances (Ahumada et al. 2013), reserve size or predator release concepts.

The species composition in Cusuco is comparable to other sites in Central- and South -America, except the absence of large predators such as jaguars and puma (Ahumada et al. 2013; Munari et al. 2011; Srbek-Araujo & Chiarello 2005). The endangered jaguar was not recorded during the study in Río Plátano either (Gonthier & Castañeda 2013). Field sign surveys undertaken in Cusuco in previous years and opportunistic findings of tracks, indicate that species such as puma and jaguar went undetected by the current study. The low detections of jaguars and pumas in Cusuco indicate that they are either present at very low numbers or not permanently residing in the park at all. When investigating distribution maps for jaguar, there is only a small patch in the North of Honduras where jaguars are considered to be residing (Caso et al. 2008). Of five Central American countries, Honduras was found to have the second lowest genetic diversity for jaguars, indicating low dispersal rates of jaguars residing in Honduras (Wultsch et al. 2016) Cusuco may function as a wildlife corridor for jaguars between the south east of Honduras and Guatemala. If this is the case, it would be important to maintain these wildlife corridors for jaguars to aid dispersal through the Mesoamerican landscape (Wultsch et al. 2016). Examining why jaguars or pumas are not residing in the park and the parks importance as a wildlife corridor will be necessary in the future. Availability of prey is unlikely to be a cause, because of high frequencies of suitable prey species in Cusuco (Table 1) (Weckel et al. 2006).

Summarised detection probabilities for field sign- and camera trap -surveys show that some species are more abundant than others. Other studies have suggested that species detection rates are related to true abundance (Trolle & Kéry 2005) and density (Mazzolli et al. 2016). The most abundant species appear to be paca, deer sp., coati, armadillo and tapir. While the low detection estimates for species such as peccary, agouti, small felid sp., tayra and grey fox indicate low abundance. Detection probabilities also suggest higher abundance of large herbivores, small herbivores and omnivores, while low abundance of carnivores. Other studies have also found that carnivores have low detection rates in comparison to other species groups (Ahumada et al. 2013; Haugaasen & Peres 2005).

4.1 Comparing survey methods

This study demonstrates how species richness did not differ according to camera trap- and field sign -surveys, while number of species detected increased slightly when methods were combined. Field sign- and camera trap surveys would be equally suitable to use to investigate species richness in Cusuco National Park. Similarly, camera trap- and field sign surveys have been found to record the same number of species (Mazzolli et al. 2016). Contrary to the current study, track surveys have been found to detected more species compared to camera traps (Silveira et al. 2003). Track surveys have also been found to be more effective recording species in time, recording the highest number of species after only 12 days, while it took 30 days for camera traps to record the same number of species (Silveira et al. 2003). Combining survey approaches increased the number of species detected to a small degree. Using both methods would be preferable, but both methods were quite effective in recording species richness. Using multiples survey approaches have been found to improve the number of species detected (Mazzolli et al. 2016).

The current study showed how estimated average detection probabilities for species varied according to camera trap- and field sign -surveys. Combining detection probabilities increased detection probabilities for all species detected by both methods. Variation in species detection rates according to survey methods have been found by other studies as well (Mazzolli et al. 2016; Nichols et al. 2008). In the current study system, camera traps were more efficient in detecting tayra, small felids, and paca. Unsurprisingly, results also show that camera traps were more effective for recording the species-groups small herbivores (dominated by paca detections) and carnivores (all are small felids).

Studies specifically designed to detect carnivores frequently use camera traps as a preferred survey approach, especially when individuals can be identified by their coat patterns (Balme et al. 2009; Spalton et al. 2006; Trolle & Kéry 2005). Even though camera traps are often used to detect carnivores, track surveys have been found to be useful to record carnivore presence as well (Beier & Cunningham 1996; Hulik et al. 2015). A study conducted in Slovakia showed higher number of detections for three carnivore species by tracks compared to camera traps (Hulik et al. 2015). Using field signs to detect carnivores may be more suitable when the species are of larger sizes, making field signs more noticeable. In the current study, the carnivores detected were only smaller felid species and smaller cat species can be difficult to identify from field signs such as tracks, dung and scats (Mazzolli & Hammer 2013; Munari et al. 2011). Field sign surveys require more experienced field personnel in comparison to camera trap surveys to identify similar species correctly (Srbek-Araujo & Chiarello 2005). Camera trapping provide a more direct indication of species presence (Srbek-Araujo & Chiarello 2005). Camera traps were also found to be more effective to detect smaller felids in the Amazon (Munari et al. 2011). Jaguarundi is easily distinguishable from camera traps, while for field signs it can be difficult to separate from margay and ocelot. In the current study, jaguarundi was certainly detected by camera traps, but because of the risk of misidentifications during field signs it had to be grouped as felid sp. for method comparison. Margay and ocelot on the other hand can be difficult to distinguish from each other from camera traps, but it is easier to consult with an expert later. Camera traps have been known to detect elusive species in neotropical forest, which may explain why camera traps performed better than field signs for species with overall low detections such as tayra and small felids (Balme et al. 2009; Trolle & Kéry 2005).

Camera traps may detect paca more effectively, because camera traps can operate at night and detect nocturnal species more easily (Srbek-Araujo & Chiarello 2005; Trolle et al. 2008). Similarly, paca was found to be more easily detected by camera traps compared to field signs in a study undertaken in the Amazon (Munari et al. 2011). Paca has also been found prefer forested areas and avoid larger trails, which would explain why camera traps placed off transect were more successful in detecting paca compared to field signs surveys on transects (Weckel et al. 2006). The smaller size of paca may have caused field signs to be more difficult to detect in comparison to for example tapir.

In addition, field surveys are often influences by field conditions, while if suitable cameras are used, field condition are rarely a problem (O'Connell et al. 2011). In a rainforest environment field signs can be washed away frequently by heavy rain, for example removing signs from species overnight (Mazzolli & Hammer 2013). In comparison, when field conditions are suitable, tracks and scats can remain present over longer periods of time (Mazzolli 2009). The study undertaken in Slovakia may have recorded more tracks compared to camera traps, because of the favourable field conditions when undertaking track surveys in snow (Hayward et al. 2002; Hulik et al. 2015). Other benefits of using camera traps are the possibly to determine sex and age, activity patterns (Trolle et al. 2008) and behaviours (Trolle & Kéry 2005). In the current study camera traps also detected other species normally not targeted by camera trap surveys such as arboreal-, small mammal- and a variety of bird -species (Appendix C).

In the current study system, field sign survey would be the preferred method to use when surveying grey fox, tapir, agouti, armadillo and coati. Unsurprisingly, field signs would also be the preferred method for detecting the species-groups omnivores (dominated by armadillo and coati detections), but also large herbivores (dominated by deer sp.). Even though these mammals have been found to be more easily detected on transects, they have also been found to be effectively surveyed by camera traps in other studies (Ahumada et al. 2013; Srbek-Araujo & Chiarello 2005; Trolle et al. 2008). Armadillo and coati might have been more easily detected by field sign surveys, because they leave obvious traces of foraging activity when searching for food. In addition, armadillo are burrowing animals and their burrows can be detected during sign surveys. In contrast, armadillo has been found to avoid trails such as transects and should have been more easily detected by cameras off transect (Weckel et al. 2006). It was a bit surprising that agouti was more easily detected by field signs than camera traps. In comparison to the results found in the current study, agouties were found to be more effectively detected by camera traps than field signs (Munari et al. 2011). The small size of agouti make signs difficult to discover and small neotropical mammals have been shown to prefer habitats with dense understory compared to open trail systems (Harmsen et al. 2010). A variety of field signs are used to record species presence and some of these may be of poor quality. Agouties were only detected for one year for field signs so possible misidentification of field signs need to be considered. Similarly, grey fox was only detected for one year and with local stray dogs also venturing inside the park, this may also have been misidentification. Tapirs have been found to be effectively monitored using field signs (Fragoso et al. 2016). On the other hand, camera traps have also been found useful to monitor tapir (Ahumada et al. 2013; Trolle et al. 2008). Larger species such as tapirs have been found to prefer man made trails compared to forested areas, which would explain why tapirs where easier to detect on transects compared to camera traps placed off transect (Harmsen et al. 2010; Weckel et al. 2006). Tapirs are possibly easier to detect for field signs because of their larger size, leaving larger tracks and dung behind.

Both field sign- and camera trap surveys would be similarly suitable to use for surveying deer sp. and peccaries. Findings match the findings of another study where peccaries showed no preferences to forest or trails (Weckel et al. 2006). Since peccaries are group animals, a benefit of using camera traps would be the possibility to record group sizes. Deer sp. species can also be difficult to distinguish from field signs and can be easier to identify from camera traps and avoid misidentifications (Mazzolli & Hammer 2013; Munari et al. 2011). Detection histories indicate that red brocket deer are more abundant in Cusuco, which in turn suggest that red brocket deer may have been misidentified as white-tailed deer. It is likely that many of the detections from field signs are red brocket deer, because white-tailed deer also prefer lower elevation in comparison.

The variation in species detection probabilities according to methods highlight the challenges of designing studies targeting mammal communities. This suggests that studies preferably need to be designed according to the focal species of interest, which have also been emphasised by other studies (Mazzolli et al. 2016). On the other hand, studies targeting communities using camera traps have become common practice to monitor study systems for conservation purposes (Ahumada et al. 2011; Ahumada et al. 2013; Rovero et al. 2014). Studies have also suggested that multiple methods are necessary for surveying multiple-species (Nichols et al. 2008). One important factor for determining survey approach in Cusuco is that field sign surveys provide long term information on mammal presence in the park and have higher detection probabilities for the endangered tapir. On the other hand, camera traps provide a more direct evidence of species presence in the park and make it easier to distinguish species from each other. Camera traps provide the possibility to individually recognize felids and even tapirs for possible capture-recapture studies (Trolle et al. 2008). Only using field sign surveys collected on transects could bias detections towards species preferably using trails and vice versa (Harmsen et al. 2010). In Cusuco it would therefore be desirable to continue to use both survey approaches to increase number of species detected and maximize species detection probabilities.

4.3 Temporal variation in the mammal community

Results show relative temporal stability in detection probability for deer, armadillo and coati with yearly oscillations, which is encouraging from a conservation point of view (Ahumada et al. 2013). Carnivores appear to show declining trends in the latest years, while the considerable declines of Baird's tapir since 2006 raises concerns (Figure 8).

Tapir declines in Cusuco national park were also demonstrated in the yearly field report from Cusuco in 2012 (Green et al. 2012) and underlined by McCann et al. (2012). Results presented herein therefore support these findings, where they expected tapir to continue to decline in Cusuco if conservation measures were not improved. Even though the park is protected, the effectivity of the protection has been questioned and the park has been termed a "paper park", because of the lack of enforcement against illegal hunting and deforestation (Martin & Blackburn 2009). Reasons for the declines are probably therefore the same as in previous years, increased illegal hunting pressure in the park as well as an increase of deforestation (McCann et al. 2012). The Brazilian tapir (*Tapirus terrestris*) has been found to be vulnerable to local extinctions because of hunting and deforestation (Trolle et al. 2008). The inaccessibility to higher elevations in cloud forests such as Cusuco has been suggested to function as a refuge for species that have lost their lowland forest habitats (Aldrich et al. 1997). Honduran cloud forests have been predicted to be lost in the future if deforestation trends continue (Martin & Blackburn 2009). A variaty of human disturbances was detected during surveys such as hunting platforms, hunting trails, animal debris, hunting huts, videos of dogs and hunters on camera traps, cleared areas, cardamom plantations, coffee plantations, and other agricultural patches were encountered during surveys and movement between camp sites (unpublished data). These findings suggest that the existing conservation management practices are not sufficiently enforced in the park to maintain carnivore and tapir populations (Ahumada et al. 2013; Martin & Blackburn 2009). Monitoring species such as tapir and carnivores can be challenging in the future, because of the difficulties of acquiring precise estimates for species with low detection rates (Ahumada et al. 2013; Haugaasen & Peres 2005; Rovero et al. 2010). Avoiding the local extinction of the endangered Baird's tapir should be a top conservation priority.

Temporal variation for camera traps were not included in the current study, because data has only been collected for three years and even five years' worth of data can be considered too short to detect temporal trends in some species (Ahumada et al. 2013). Both paca and small felid sp. were better detected by camera traps, consequently cams may be a more suitable method to investigate temporal variations for these species in the future. An extension to the current study would be to include yearly covariates to the occupancy model, which will improve the understanding of possible extinctions and colonisations of species, taking into account if species were detected in previous years or not (Ahumada et al. 2013). It will be necessary to develop effective methods to monitor levels of human disturbances in the park to investigate possible effects on the mammal community to aid conservation (Ahumada et al. 2011).

4.4 Survey approach and design considerations

The current study was limited by small sample size (<15) for some of the species. This was reflected by the large confidence intervals for some species for certain years, making detection probability estimates unprecise (Trolle & Kéry 2005).There was also no statistical test undertaken to investigate the significant difference of species detection probabilities according to methods, which would have been preferable to carry out. Relative differences still provide us with a baseline for designing future studies. Covariates such as habitat preferences and human disturbances are also probably affecting detection probabilities as well as survey approach. Effects of covariates were not included in the current study, but should be investigated in future studies. A different camera trap survey design should be considered to monitor large mammal communities in Cusuco in the future. The current camera trap survey design was designed to investigate the effects of transect use on large mammals and to calculate abundance using the Random Encounter Model (REM) (Reid 2016a). Using Camera trap surveys can be more time efficient than field sign surveys, since time is only spent placing cameras and maintaining them during the study period (Srbek-Araujo & Chiarello 2005; Trolle et al. 2008). In Cusuco the current camera trap design of placing and retrieving 6 cameras up to 300m off transect was very time consuming, due to inaccessibility off transect path. Since 6 cameras were necessary on each transects and a limited number of cameras were available, the cams had to be placed sequentially through the park and could only be left in the field for 3 days. The disturbance caused when placing a camera may reduce detections within the first days, because animals may avoid human disturbed areas (O'Connell et al. 2011). Leaving cameras in the field for only three days, yielded approximately 350 camera trap nights per year in 2015 and 2016, which in comparison to other camera trap surveys would be considered very low survey effort. Other surveys rarely yield less than 1000 camera trap days per year (Tobler et al. 2008) or cameras are normally active for 30-60 days (Trolle et al. 2008). The current study would only come close to 1000 trap nights after 3 years of surveying. Fewer cameras were placed in 2014, but they were left in the field for longer (up to 28 days). More species were detected in 2014 as well as a higher number of species were detected for most species (Appendix C). This suggests that that such an approach would be more beneficial to monitor species in Cusuco. For maximum camera trapping efforts during one season, the cams would need to be placed during the first opportunity when camps open and retrieved at the latest chance before camps close. If cams were active for a longer period of time, it would be possible to further investigate detection probabilities by additionally using time-to event analysis (Bischof et al. 2014). The necessity of high survey effort and the use of multiple survey approaches to monitor large mammal species have been highlighted by another study (Munari et al. 2011).

The proximity of cameras also prevents the independence assumption to be met, which is necessary to acquire true occupancy estimates (O'Connell & Bailey 2011). With the right survey design, occupancy estimates can provide useful knowledge on species distributions in space and time (Nichols et al. 2008). Occupancy is probably the most suitable measure to monitor large mammal communities in Cusuco, because of the low number of detections, elusive species and difficult access in rainforests. Occupancy can also in some situations be used as a proxy for abundance (MacKenzie 2006). Most monitoring studies spread cameras across the study area with a minimum of 1km space apart to meet the independence assumption (Bischof et al. 2014; Trolle et al. 2008). The suitability of a grid configuration system should be investigated, since this approach have been used successfully for other studies (Bischof et al. 2014; Mazzolli et al. 2016; Trolle & Kéry 2005). Camera traps and field sign surveys should be undertaken within the same grid to make detections more compatible (Mazzolli et al. 2016). This approach would also provide the necessary framework to include both methods in the same occupancy model, improving accuracy of occupancy and detection probability estimates (R. Bischof, personal communication, Spring, 2017). The importance of integrating different sampling methods in the same statistical framework has been highlighted by Munari et al. (2011). In the future, correlations between detecting a species by one study given that it was detected by the other should be investigated as well (Bischof et al. 2014).

Preferably, cameras would be placed on transect as well to avoid bias towards species preferring off trail habitats (Harmsen et al. 2010; Munari et al. 2011; Weckel et al. 2006). The risk of cameras getting stolen along transects may prevent this from being possible. Cameras could possibly be locked to large trees along transects, but more management of changing memory cards will be necessary since transects are used frequently by other field personnel for a variety of surveys (O'Connell et al. 2011; Rovero et al. 2010). A camera trap design similar to the Team Network protocol should be considered (TEAM Network 2011). By following such a protocol, findings can be compared across multiple studies in Central America. These data have already provided information on effects of human disturbances (Ahumada et al. 2011; Ahumada et al. 2013; Rovero et al. 2014), true occupancy estimates (Ahumada et al. 2013), temporal dynamics (Ahumada et al. 2013), species richness estimates (Rovero et al. 2014), wildlife picture index (Ahumada et al. 2013), habitat preference (Rovero et al. 2014), comparison across sites (Ahumada et al. 2011) and species accumulation curves (Rovero et al. 2014). Even though camera trapping is more expensive, studies often find the benefits override the disadvantage of high initial costs (Balme et al. 2009; Silveira et al. 2003; Srbek-Araujo & Chiarello 2005; Trolle et al. 2008).

4.5 Conclusion

This study provides valuable information to be able to undertake effective monitoring programs for large mammal communities using camera trap surveys and field sign surveys in Cusuco National Park. Species are decreasing at an alarming rate in many study systems and detecting possible declines in time to implement conservation measures are crucial. Combining multiple survey techniques are preferred in most situations, if time and expenses allow for it.

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6. Appendices

Appendix A

Map of Cusuco National park including elevation gradient and the seven main camp sites. The transparent mask indicates the area outside the park (Green et al. 2012).

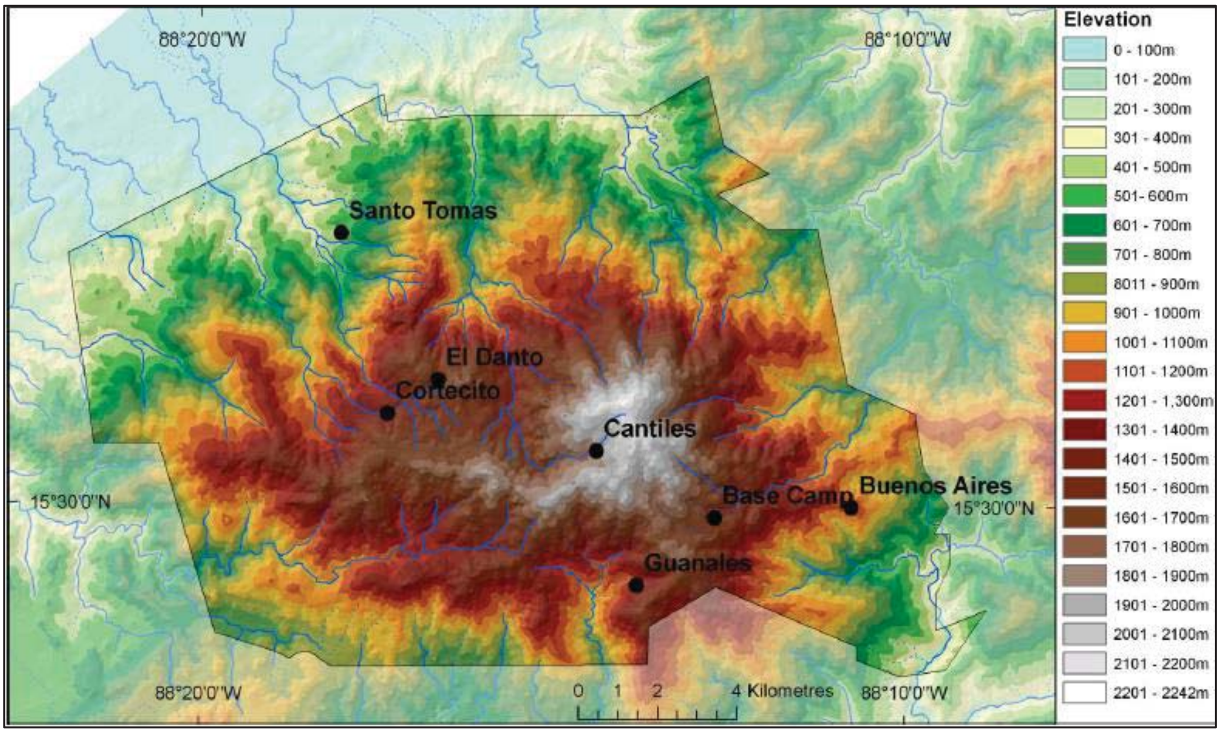

Figure A1: Map of Cusuco National park including elevation gradient and the seven main camp sites. The transparent mask indicates the area outside the park.

Appendix B

Camera settings:

- Mode = Video
- Image size = 8M pixel
- Image format = Full screen
- Capture number = 1
- IR LED Control = Medium
- Camera name = Leave as Input
- Video size = 1280*720
- Video length = 20s
- Interval = 1 minute
- Sensor level = Auto
- NV Shutter = High
- Camera mode = 24hrs
- Format = Deletes everything on SD card (not in use)
- Time Stamp = On
- Set Clock = Local time and date
- Field Scan = Off
- Coordinate Input = Off
- Set video sound = On
- Default = Cancel
- Version = BS683BWY*05212

Appendix C

Table C1: Overview of camera trap and field sign detections from 2014 to 2016

	Camera trap detections			Field sign detections			
Scientific name	**2014**	**2015**	**2016**	**2014**	**2015**	**2016**	**Total**
Tapirus bairdii	9	1	0	24	29	1	64
Mazama americana	17	11	8	40	32	12	88
Odocoileus virginianus	1	0	0	0	0	0	33
Cuniculus paca	29	30	15	0	3	8	85
Pecari tajacu	19	9	0	1	13	4	46
Dasyprocta punctata	1	0	2	37	0	0	40
Dasypus novemcinctus	4	0	0	10	20	19	53
Herpailurus yaguarondi	0	1	0	0	0	0	1
Leopardus pardalis	0	1	0	0	1	0	2
Leopardus wiedii	4	6	1	0	0	1	12
Nasua narica	13	11	6	42	11	19	102
Potos flavus	6	1	1	6	4	4	22
Eira Barbara	3	0	0	0	0	0	3
Mustela frenata	4	0	0	0	0	0	4
Urocyon cinereoargenteus	0	0	0	3	0	0	3
Didelphis sp	14	8	11	11	1	2	47
Mephitis sp.	0	0	0	0	0	2	2
Alouatta palliata	0	0	0	4	4	0	8
Sylvilagus brasiliensis	0	0	0	0	9	0	9
Sciurus sp.	47	42	34	0	0	0	123
Cricetidae sp.	43	80	71	0	0	0	194
	0	5	1	0	3	0	9
Canis lupus familiaris	0	2	2	0	0	0	4
Homo sapiens	0	1	4	0	0	0	5
Penelopina nigra	NA	15	21	NA	NA	NA	36
Crax rubra	NA	2	0	NA	NA	NA	2
Odontophorus guttatus	NA	4	10	NA	NA	NA	14
Crypturellus sp.	NA	3	4	NA	NA	NA	7
Geotrygon albifacies	NA	36	79	NA	NA	NA	115
Catharus mexicanus	NA	22	14	NA	NA	NA	36
Arremon brunneinucha	NA	60	43	NA	NA	NA	103
Micrastur ruficollis	NA	1	0	NA	NA	NA	1
Lampornis viridipallens	NA	0	1	NA	NA	NA	1
Formicarius moniliger	NA	0	1	NA	NA	NA	1
Momotidae sp.	NA	0	4	NA	NA	NA	4
Catharus frantzii	NA	2	0	NA	NA	NA	2
Grallaria guatimalensis	NA	2	1	NA	NA	NA	3
Sclerurus guatemalensis	NA	2	0	NA	NA	NA	2
Turdidae sp.	NA	1	1	NA	NA	NA	2
Henicorhina leucosticta	NA	0	1	NA	NA	NA	1
Troglodytidae sp.	NA	1	0	NA	NA	NA	1
Unknown bird sp.	NA	5	11	NA	NA	NA	16
Large Herbivore	46	21	8	65	74	17	231
Small Herbivore	30	30	15	37	3	8	123
Omnivore	34	19	17	66	32	42	161
Carnivore	4	8	1	0	1	1	15

Appendix D

Table D1: Overview of field sign detections from 2006 to 2016

Animal name	2006	2007	2008	2009	2010	2011	2012	2013	2014	2015	2016	Total
Tapirus bairdii	162	44	153	310	132	22	27	13	24	29	1	917
Mazama americana	10	1	91	38	95	98	169	79	40	32	12	633
Odocoileus virginianus	1	2	5	16	18	3	8	5	0	0	0	90
Cuniculus paca	7	2	22	54	77	69	73	60	0	3	8	375
Pecari tajacu	0	0	0	0	0	7	9	0	1	13	4	34
Dasyprocta punctata	0	0	4	0	0	0	0	1	37	0	0	42
Dasypus novemcinctus	3	1	89	36	47	17	37	33	10	20	19	312
Herpailurus yaguarondi	0	0	0	3	0	0	5	2	0	0	0	10
Leopardus pardalis	0	0	4	5	5	1	10	3	0	1	0	29
Leopardus wiedii	0	0	0	0	0	0	2	0	0	0	1	3
Puma concolor	0	0	1	1	0	0	0	0	0	0	0	2
Nasua narica	0	3	119	115	100	43	104	35	42	11	19	591
Potos flavus	0	0	1	3	4	5	10	3	6	4	4	40
Procyon lotor	1	0	9	4	7	3	4	1	0	0	0	29
Bassariscus sumichrasti	0	0	0	0	0	1	0	0	0	0	0	1
Eira barbara	0	0	1	5	6	1	1	2	0	0	0	16
Urocyon cinereoargenteus	0	0	0	0	9	0	0	0	3	0	0	12
Didelphis sp.	0	0	22	12	15	6	17	18	11	1	2	104
Mephitis sp.	0	0	0	8	18	3	6	6	0	0	2	43
Alouatta palliata	82	11	12	16	9	7	10	4	4	4	0	159
Ateles geoffroyi	2	0	0	0	0	0	0	0	0	0	0	2
Cebus capucinus	1	0	0	0	0	0	0	1	0	0	0	2
Coendou mexicanus	0	0	2	0	0	0	0	0	0	0	0	2
Sylvilagus brasiliensis	0	0	0	0	0	0	0	0	0	9	0	9
Sciurus sp.	3	1	0	0	0	0	0	0	0	0	0	4
Orthogeomys sp	0	0	3	4	0	0	0	5	0	0	1	13
Deer sp.	11	3	96	54	113	101	177	84	40	32	12	723
Large Herbivore	173	47	249	364	245	130	213	97	65	74	17	1674
Small Herbivore	7	2	26	54	77	69	73	61	37	3	8	417
Omnivore	5	4	240	180	202	73	169	95	66	32	42	1108
Carnivore	0	0	5	9	5	0	17	5	0	1	1	43
Hunting platform	3	0	0	0	5	7	9	8	5	5	0	42
Unknown sp.	5	3	39	9	8	0	0	0	0	3	0	67

Appendix E

Table E1: Summary of estimated means and critical intervals from the posterior distributions. Detection probability (p), local site occupancy (Ψ) and local transect occupancy ($\emptyset$) for species with a minimum of 5 detections for one of three years.

Mammal Species	**Detection probability; $mean\ p\ (CI)$**			**Occupancy site; $mean\ \Psi$ (CI) psiS**			**Occupancy transect; $mean\ \emptyset\ (CI)$ psiT**		
	$p14$	$p15$	$p16$	$\Psi14$	$\Psi15$	$\Psi16$	$\emptyset14$	$\emptyset15$	$\emptyset16$
Cuniculus paca	0.099 (0.061-0.145)	0.265 (0.117-0.437)	0.229 (0.080-0.437)	0.683 (0.396-0.961)	0.279 (0.139-0.541)	0.392 (0.138-0.859)	0.813 (0.506-0.993)	0.849 (0.592-0.994)	0.557 (0.249-0.941)
Mazama americana	0.102 (0.048-0.172)	0.090 (0.018-0.279)	0.105 (0.028-0.267)	0.424 (0.130- 0.887	0.505 (0.112-0.967)	0.592 (0.176-0.975	0.606 (0.198-0.980)	0.644 (0.254- 0.980)	0.510 (0.182-0.942)
Nasua narica	0.049 (0.020-0.091)	0.092 (0.018-0.286)	0.098 (0.018-0.305)	0.695 (0.328-0.983)	0.508 (0.106-0.969)	0.491 (0.102-0.967)	0.726 (0.365-0.986)	0.567 (0.190-0.968)	0.617 (0.222-0.978)
Pecari tajacu	0.049 (0.020-0.092)	0.097 (0.019- 0.290)	DD	0.696 (0.323- 0.984)	0.502 (0.103-0.968)	DD	0.726 (0.366-0.987)	0.546 (0.181-0.962)	DD
Didelphis.sp.	0.052 (0.021-0.099)	0.167 (0.025- 0.451)	0.246 (0.090- 0.475)	0.662 (0.298-0.977)	0.270 (0.045-0.855)	0.560 (0.193-0.960)	0.740 (0.379-0.988)	0.637 (0.229-0.981)	0.281 (0.093-0.593)
Tapirus bairdii	0.047 (0.011- 0.115)	DD	DD	0.527 (0.163-0.956)	DD	DD	0.680 (0.253- 0.986)	DD	DD
Leopardus wiedii	DD	0.097 (0.017 0.311)	DD	DD	0.464 (0.092-0.960)	DD	DD	0.599 (0.212 0.975)	DD
Large Herbivore	0.085 (0.049- 0.129)	0.092 (0.029- 0.231)	0.106 (0.027- 0.271)	0.675 (0.386- 0.960)	0.615 (0.204- 0.978)	0.592 (0.172- 0.978)	0.792 (0.471-0.992)	0.653 (0.310- 0.977)	0.510 (0.176- 0.948)
Small herbivore	0.091 (0.057-0.132)	0.265 (0.116-0.440)	0.266 (0.104-0.470)	0.739 (0.451-0.976)	0.286 (0.142-0.571)	0.350 (0.136-0.758)	0.837 (0.557-0.994)	0.848 (0.589-0.994)	0.600 (0.293-0.953)
Omnivore	0.083 (0.053-0.119)	0.107 (0.031-0.270)	0.169 (0.066- 0.337)	0.819 (0.551-0.991)	0.480 (0.143-0.957)	0.589 (0.242- 0.968)	0.832 (0.568-0.992)	0.797 (0.472-0.992)	0.555 (0.275- 0.913)
Carnivore	0.034 (0.007- 0.088)	0.086 (0.020- 0.257)	DD	0.619 (0.200- 0.979)	0.503 (0.122- 0.967)	DD	0.675 (0.246- 0.987)	0.710 (0.331- 0.987)	DD
Arboreal	0.053 (0.003- 0.187)	DD	DD	0.444 (0.047- 0.962)	DD	DD	0.428 (0.045- 0.958)	DD	DD
Squirrel sp.	0.165 (0.118- 0.217)	0.227 (0.115- 0.361)	0.237 (0.115- 0.384)	0.818 (0.507- 0.994)	0.447 (0.255- 0.772)	0.420 (0.233- 0.233)	0.668 (0.386- 0.959)	0.897 (0.696- 0.997)	0.909 (0.715- 0.997)
Geotrygon albifacies	NA	0.155 (0.075-0.281)	NA	NA	0.595 (0.297-0.960)	NA	NA	0.834 (0.587- 0.991)	NA
Catharus mexicanus	NA	0.573 (0.330-0.780)	0.140 (0.038-0.333)	NA	0.253 (0.075-0.502)	0.517 (0.143-0.965)	NA	0.302 (0.095-0.711)	0.486 (0.172- 0.924)
Arremon brunneinucha	NA	NA	0.224 (0.075- 0.385)	NA	NA	0.427 (0.211- 0.814)	NA	NA	0.809 (0.537- 0.989)
Penelopina nigra	NA	0.108 (0.037-0.259)	0.108 (0.043-0.223)	NA	0.535 (0.179- 0.966)	0.710 (0.315-0.987)	NA	0.758 (0.430- 0.988)	0.640 (0.327-0.966)
Odontophorus guttatus	NA	0.136 (0.007- 0.519)	0.101 (0.016-0.334)	NA	0.348 (0.025-0.939)	0.441 (0.077-0.958)	NA	0.450 (0.068-0.958)	0.640 (0.226-0.982)

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