Occupancy of the Sooty Ant-Tanager (*Habia gutturalis*) and White-Mantled Barbet (*Capito hypoleucus*) in fragmented forests of the Central Andes in Colombia

Ocupación del habia ceniza (*Habia gutturalis*) y el torito capiblanco (*Capito hypoleucus*) en bosques fragmentados de la Cordillera Central en Colombia

Jefry S. Betancur[1], Andrea Morales-Rozo[1,2] & Juan L. Parra[1]

[1] Grupo de Ecología y Evolución de Vertebrados, Instituto de Biología, Universidad de Antioquia, Medellín, Colombia
[2] Grupo de Investigación ECOTONOS, Programa de Biología, Facultad de Ciencias Básicas e Ingeniería, Universidad de los Llanos, Sede Barcelona km 12 vía Puerto López, Villavicencio, Meta
✉ styf17@gmail.com, amoralesrozo@unillanos.edu.co, juanl.parra@udea.edu.co

Abstract

The Sooty Ant-Tanager (*Habia gutturalis*) and White-Mantled Barbet (*Capito hypoleucus*) are endangered and endemic birds of Colombia. Both species have small geographic ranges and presumably low population sizes possibly due to habitat destruction and fragmentation. In order to estimate the effects of landscape features on the occupancy of both species, we sampled a variety of landscape configurations within the buffer zones of two hydroelectric impoundments in the Central Andes of Colombia and applied occupancy models to estimate the proportion of area occupied as a function of these covariates. We surveyed 35 point-counts in each hydroelectric impoundment, between June and July of 2014 and 2015. We used single-season models to estimate occupancy while recognizing imperfect detection. Mean occupancy estimates in the study area were similar for both species (0.61 SD=0.33 for the Sooty Ant-Tanager and 0.63 SD=0.25 for the White-Mantled). Nonetheless, occupancy probability within the study area was very different between them. The best model for the Sooty Ant -Tanager indicated a decrease in occupancy with elevation, whereas the top model for the White-Mantled Barbet indicated an increase in occupancy with distance from streams. Detection probabilities were similar for both species (>0.4) and declined significantly during the second year. Our results provide quantitative guidelines that can be used to evaluate and monitor the state of these populations on the short and long term.

Key words: birds, Colombia, conservation, detectability, hydroelectric, dam

Resumen

El habia ceniza (*Habia gutturalis*) y el torito capiblanco (*Capito hypoleucus*) son aves amenazadas y endémicas de Colombia. Ambas especies tienen rangos geográficos y tamaños poblacionales pequeños posiblemente como resultado de la destrucción y fragmentación de sus hábitats. Con el fin de obtener estimados cuantitativos de los efectos de características del paisaje sobre la ocupación de ambas especies, muestreamos una variedad de configuraciones del paisaje al interior de las zonas de amortiguamiento de dos hidroeléctricas en la cordillera Central de los Andes de Colombia y empleamos modelos de ocupación para estimar la proporción del área ocupada en función de estas covariables. Realizamos 35 puntos de conteo en cada zona de amortiguamiento entre junio y julio del 2014 y 2015. Utilizamos modelos de ocupación de una sola temporada para estimar la ocupación reconociendo la detección imperfecta. Los promedios de ocupación para ambas especies en el área de estudio fueron similares (0.61 SD=0.33 para el habia ceniza y 0.63 SD=0.25 para el torito capiblanco). Sin embargo, la distribución de la ocupación al interior del área de estudio fue muy diferente entre ellas. El mejor modelo para la habia ceniza propone que su ocupación disminuye con la elevación, mientras que el mejor modelo para la ocupación del torito capiblanco propone un aumento en la ocupación con la distancia a quebradas. Las probabilidades de detección fueron similares para ambas especies (<0.4) y declinaron significativamente durante el segundo año. Nuestros resultados proveen lineamientos cuantitativos para evaluar y monitorear el estado de estas poblaciones a corto y largo plazo.

Palabras clave: aves, Colombia, conservación, detectabilidad, hidroeléctricas, represa

Introduction

The Magdalena valley moist forest region lies within the interandean Valley of the Central and Eastern Cordilleras of Colombia and holds a great variety of ecosystems, including 1.5 million hectares of forests and natural savannah. This region includes more than 630 bird species and sustains a substantial amount of threatened birds in Colombia, making it a priority for conservation (*e.g.*, ~60% of all threatened species in Colombia, Renjifo *et al.* 2002, Renjifo *et al.* 2014, WCS 2019). Nonetheless, the Magdalena Valley moist forest region has been under threat for decades with a high deforestation rate that started during the 1960's and 1970's as a consequence of the human colonization front that replaced natural vegetation cover by cropland and pastures along the Magdalena river; currently the rate of deforestation in this region is estimated to be 81000 ha per year (Idárraga *et al.* 2016). The key drivers of deforestation and biodiversity loss in this region include agriculture expansion, illicit crops, development of infrastructure, illegal wood extraction, and mining (González *et al.* 2011).

An increasing source of fragmentation, especially within the topographically complex Andean region, is the construction of hydroelectric impoundments (Terborgh *et al.* 2001, Finer & Jenkins 2012, Anderson *et al.* 2018), particularly in the departments of Caldas and Antioquia (Múnera 2016). Although the economic benefits of impoundments are eminent, their impacts on biodiversity (Kupferberg *et al.* 2012, Pandit & Grumbine 2012, Ziv *et al.* 2012) and the altered ecosystem functions (Finer & Jenkins 2012) have been largely ignored. The two most eminent indirect effects of fragmentation caused by impoundments and most other fragmentation sources are an overall decrease in patch area and an increase in isolation. Nonetheless, these two entail additional indirect effects, for example, a decrease in patch area can influence trophic interactions through an increase in the relative edge length, which not only changes the microclimate but also exposes individuals to predation and parasitism (Wilcove 1985). Various studies have documented an increase in nest predation after forest fragmentation, causing significant reproductive loss for birds (Simberloff &Wilson 1970, Diamond 1972, Lahti 2001, Feeley & Terborgh 2006). Other studies have documented positive responses from species associated to dense second growth and forest edges (Thiollay 1997).

Studies about the impacts of deforestation on birds have shown mixed results. For instance, understory insectivorous birds are largely negatively affected (Stratford & Stouffer 1999, Powell *et al.* 2015), and few studies have shown mixed or even positive responses for other birds (Boyle & Sigel 2015). Initiatives to mitigate the effects of fragmentation and manage landscapes affected by it, including hydroelectric dams in tropical regions, require information about the responses of biodiversity to changes in landcover. An important input to monitoring programs and conservation assessments is the proportion of area occupied by an organism, also known as occupancy (Mackenzie & Reardon 2013). In this study, we provide information on the occupancy patterns of two endangered and endemic bird species in fragmented forests in the Central Andes of Colombia: *Habia gutturalis* and *Capito hypoleucus*, considered near threatened (NT) and vulnerable (VU) respectively (Renjifo *et al.* 2014, BirdLife International 2016a, 2016b, Múnera 2016). The two bird species studied have restricted ranges within the northern Colombian Andes (*H. gutturalis* ~117,000 km^2, *C. hypoleucus* 2,000 km^2). The Sooty Ant-Tanager is found in the Magdalena and Sinú Valley moist forests usually between 100 – 1100 m, where it inhabits humid-forests and is associated with streams, landslides,

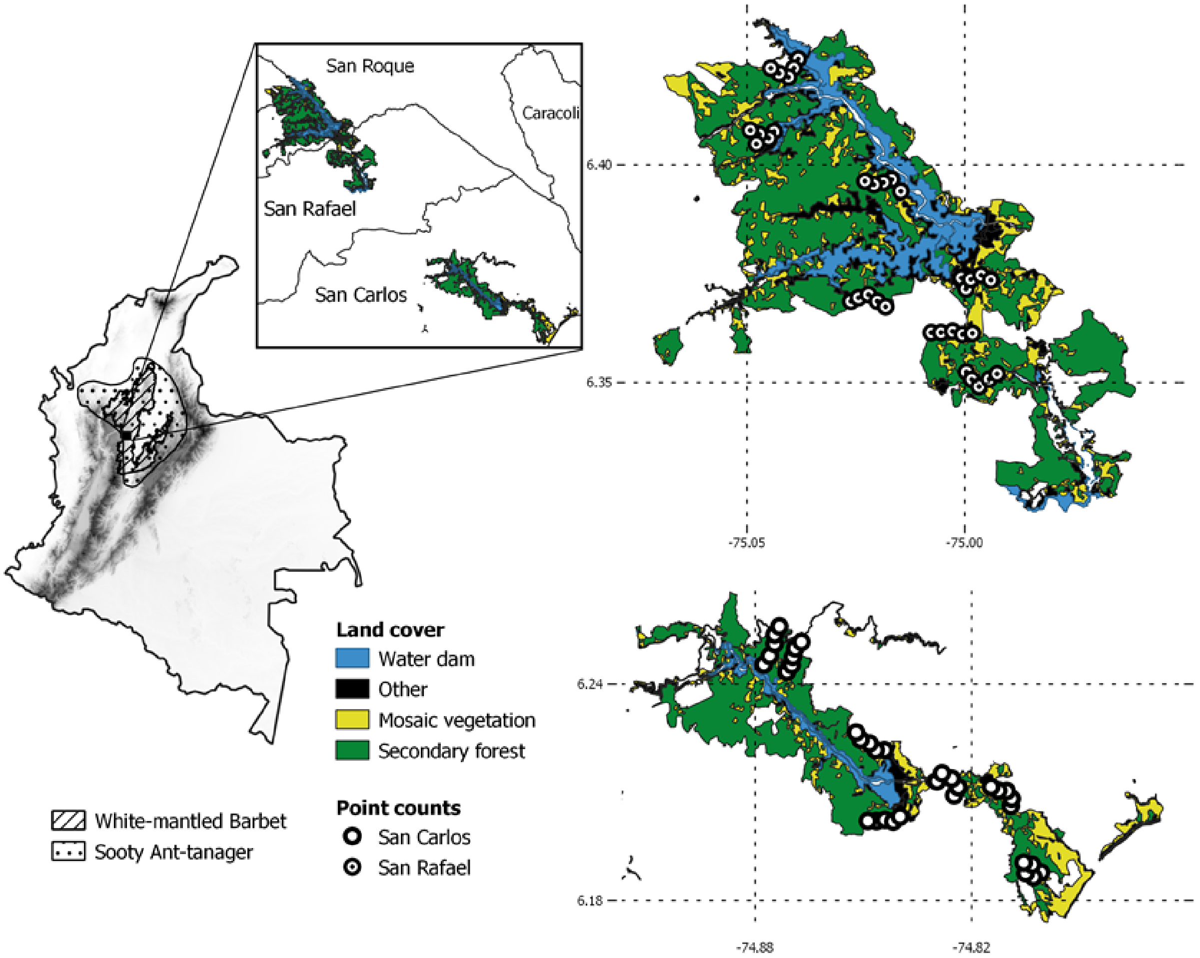


Figure 1. Location of the study area in Colombia with respect to the geographic ranges of the two endemic birds. Maps to the right represent the buffer zones of each impoundment (San Lorenzo above and Punchiná below) with the location of the survey sites.

and forest clearings (Willis 1972, Renjifo *et al.* 2002, Renjifo *et al.* 2016). The White-Mantled Barbet is distributed in the northern part of the Central Cordillera and its eastern flanks where it makes use of forests, secondary growth and other lightly disturbed areas from 400 – 2000 m, but usually above 1000 m (Laverde *et al.* 2005, Renjifo *et al.* 2016). Even though both species have been considered rare (Hilty & Brown 2001), they have also been identified as locally common by some authors (Stiles *et al.* 1999) indicating certain tolerance to human disturbance.

Our goal was to evaluate the potential impacts of forest fragmentation, nest predation, and other environmental covariates on occupancy probability for these two species within the buffer zones of two hydroelectric impoundments. We evaluated whether heterogeneity in occupancy across sites was related to the following variables: percentage of forest and crops within a buffer zone, elevation, nest predation intensity and distance to water bodies. We expected both species to tolerate some degree of deforestation and to be negatively impacted by closeness to impoundments and high predation rates; the Sooty Ant-Tanager is expected to be positively affected by nearness to streams, while the White-Mantled Barbet is expected to thrive in highly forested areas. Simultaneously, we asked if heterogeneity in detection was related to weather variables (precipitation and temperature), time of day and season (2014 or 2015). We used single season occupancy models to evaluate these relationships accounting for imperfect detection. We emphasize the use of occupancy modeling in studies of biodiversity and discuss the implications of our results for the conservation of these two endangered birds.

Methods

Study area.- The study was conducted in fragmented secondary forests within the buffer zones of two hydroelectric power plants located in the department of Antioquia in the Central Andes of Colombia: one in the municipality of San Carlos (Punchiná), and the other in the municipalities of San Rafael, Alejandría, Santo Domingo and San Roque (San Lorenzo; Fig. 1). These protected areas encompass ~80 km^2 (ISAGEN 2008) of transitional forest from premontane to tropical wet forests (Holdridge 1947), a key elevation band (580-1420 m) characterized by high levels of fragmentation (Cardona *et al*. 2011, Fig.1).

Sampling.– Our sampling units were point counts defined as circles of 125 m radius, within which birds were registered during 30 minutes. We defined the maximum radius, as the longest distance at which there was a record of any of the two species. Point counts were distributed along transects of ~1 km each. Transects were located near trails and separated by more than 2 km from each other. Each transect consisted of five point-counts spaced at 250 m intervals along a route. A total of 70-point counts (14 transects) were located within the study area, subject to security and transportation restrictions, with the intention to include a broad range of landscape configurations. The landscape features were based on a remotely-sensed Landsat 2013 image (Appendix 1). We surveyed 35 sites in the buffer zone of each hydroelectric impoundment between June and July of 2014 and 2015. One of the occupancy modeling assumptions is that detection histories are independent across sites (Mackenzie *et al*. 2002). In order to quantitatively assess this assumption, we performed spatial correlograms on the residuals of a logistic regression between detections and occupancy covariates.

We conducted all surveys from sunrise (06:00) until 10:00, waiting silently for three minutes before the beginning of observations. For each target species, we registered all individuals seen or heard during a 30-min period, and we recorded each survey with an omnidirectional microphone Sennheiser ME62. Two observers (AMR in 2014 and JSB in 2015) conducted the surveys for two consecutive days. During a survey, we scored the distance to each detection with prior training using a range finder (Nikon 8397 ACULON). We also quantified the number of individuals of each species and if the detection was visual or auditory. The two species are medium sized birds (~20 cm) that are at least seasonally vocal with distinct and easily recognizable songs and calls (Willis 1972, Hilty & Brown 2001, Laverde *et al*. 2005). In total we conducted 70-point counts with two repetitions during the course of one year.

Data analysis.– To obtain maximum-likelihood estimates of occupancy and detection for each species, we used a single-season occupancy model (Mackenzie *et al*. 2002). This model assumes that occupancy states do not change during the study, and in our particular case we assume that occupancy status (presence or absence at a site) is constant across one year (June 2014-July 2015). We provide evidence that this is a reasonable assumption by analyzing the data separately for surveys made each year. It is important to remember that this assumption refers to the state of the site and not to the

Table 1. Results of single-season occupancy models for the White-Mantled Barbet (*Capito hypoleucus*). Model results are presented in descending order based on AICc. Model notation includes the covariates used within parentheses for occupancy (Ψ) and detection (ρ). Points within parentheses indicate models without covariates. The five models where occupancy is constant (.) were evaluated first to identify the most likely covariates for detection. Covariates include the shortest straight distance to streams (Dist.str), the shortest straight distance to impoundments (Dist. imp), time of the survey (H.max), the average temperature during the survey (Temp), and the precipitation of the previous day (Prec).

Model	Number of parameters	AICc	ΔAICc	AICw
$\Psi(Dist.str)\rho(year)$	4	231.13	0	0.608
$\Psi(.)\rho(year)$	3	234.94	3.81	0.08
$\Psi(Elev)\rho(year)$	4	235.24	4.11	0.078
$\Psi(Forest)\rho(year)$	4	235.31	4.18	0.075
$\Psi(Mosaic)\rho(year)$	4	236.19	5.06	0.049
$\Psi(Dist.imp)\rho(year)$	4	237.03	5.90	0.032
$\Psi(Predation)\rho(year)$	4	237.13	6.00	0.029
$\Psi(.)\rho(H.max)$	3	237.39	6.26	0.023
$\Psi(.)\rho(Temp)$	3	237.44	6.31	0.023
$\Psi(.)\rho(.)$	2	237.48	6.35	0.02
$\Psi(.)\rho(Prec)$	3	238.62	7.49	0.013

presence of a particular individual at a site (Mackenzie *et al.* 2018). All analyses were carried out the R-package unmarked v. 0.10-2 (Kéry *et al.* 2005, Fiske & Chandler 2011).

We fitted models with increasing levels of complexity and used Akaike's information criterion to evaluate among them. We used the year, the time of the survey, the average temperature during the survey (Temp), and the precipitation of the previous day (Prec) as detection covariates. Year was included as a detection covariate since the person in charge of observations changed from 2014 (AMR) to 2015 (JSB). The other covariates have been shown to affect detection probability of birds as they relate to foraging and vocal activities (Farnsworth *et al.* 2002, Royle *et al.* 2005). Climate data were derived from two local weather stations: San Lorenzo (1252 m) and Porto Belo (775 m), each point count received the climate data of the nearest station.

We first evaluated among detection models assuming constant occupancy for each species, and after choosing the best model for detection, we evaluated six models using the following covariates for occupancy (Table 1): the percentage of the area of fragmented forest (Forest) in a 125m radius, the percentage of the area of crop mosaic cover (Mosaic) in a 125m radius (Appendix 1), elevation (Elev), the shortest straight distance to impoundments (Dist. imp), the shortest straight distance to streams (Dist.str), and nest predation intensity (Predation). This last covariate was estimated based on the results of a nest predation experiment with artificial open cup nests that do not resemble the nests of either of the studied species. The purpose of this covariate was a general measurement of predation (not predation for a particular species) and how this changed under different landscape configurations. The experiment estimated nest predation by quantifying the number of artificial nests predated at each site. Five round cup nests were exposed to heights between 0.5 and 1.5 m not less than 10 m distance between them within every sampling unit (point count). Each nest contained a common quail egg (*Coturnix coturnix*) and an imitation plasticine egg (Appendix 2). Each nest was revised once per day, for a total of two days per nest. We measured the frequency of predation using the following equation:

$$Predationfrequency = \frac{Nestspredated}{totalnestsxNumberofdaysexposed}$$

A nest was considered predated if their eggs had laceration marks or an egg was missing. The frequency of predation was measured each year and the maximum frequency of predation for a

Table 2. Summary of the single-season occupancy models fit for the Sooty Ant-Tanager (*Habia gutturalis*). Model results are presented in descending order based on AICc. Model notation as explained in Table 1.

Model	Number of parameters	AICc	ΔAICc	AICw
$\psi(Elevation)\rho(year + Prec)$	5	243.38	0.00	≈1
$\psi(Dist.imp)\rho(year + Prec)$	5	261.93	18.56	<0.001
$\psi(.)\rho(year + Prec)$	4	262.84	19.78	<0.001
$\psi(.)\rho(year + +H.max)$	5	263.09	19.71	<0.001
$\psi(Forest)\rho(year + Prec)$	5	263.50	20.13	<0.001
$\psi(.)\rho(year)$	5	263.88	21.08	<0.001
$\psi(Predation)\rho(year + Prec)$	5	264.07	20.69	<0.001
$\psi(Dist.str)\rho(year + Prec)$	3	264.73	21.36	<0.001
$\psi(Mosaic)\rho(year + Prec)$	5	265.12	21.74	<0.001
$\psi(.)\rho(Prec)$	3	266.89	24.09	<0.001
$\psi(.)\rho(H.max)$	3	269.04	26.23	<0.001
$\psi(.)\rho(.)$	2	269.75	27.14	<0.001
$\psi(.)\rho(Temp)$	3	270.47	27.67	<0.001

site was chosen as an indirect measure of predation pressure for each point count.

All covariates were standardized prior to analysis (standardized value = value - mean/standard deviation) except the time of start of the point count, which was transformed to angular distances (the arcsine of the root square of the time as a proportion). We selected the best models based on the lowest AICc. We also evaluated that variability in our data was higher than expected under our given model (overdispersion) by using the Pearson statistic as a goodness of fit test using 1000 parametric bootstraped samples with the function mb.gof.test from the AICcmodavg package (Mackenzie & Bailey 2004). We provide estimates of model parameters (detection and occupancy) and their associated statistical uncertainty. When the 95% confidence interval of a coefficient of a covariate did not include 0, we plot occupancy as a function of the covariate and we also generated maps of the probability of occupancy within the study area for both species (Chandler *et al.* 2015). In order to represent the average probability of occupancy within the buffer zones of each impoundment, we used the best model for each species, predicted occupancy for all sample units within each region, and then computed their mean and standard deviation.

Results

The Sooty Ant-Tanager (*H. gutturalis*) was detected in 33 out of 70 sites (73 detections in 280 visits) for a naïve occupancy estimate of 0.47. Most detections were auditory (56.16%) in the undergrowth of secondary forests. Birds were observed usually in pairs or small family groups (3 -4) in the understory, sometimes following swarms of army ants (four out of 32 visual detections). The White-Mantled Barbet (*C. hypoleucus*) was detected in 28 sites (44 detections in 280 visits) for a naïve occupancy estimate of 0.4. Most detections were also auditory (63.63%) but in the canopy and subcanopy of secondary forest. Birds of this species were observed in groups from two to three individuals foraging in the canopy.

Results of analyses of detection histories by year (June 2014 vs July 2015) indicated that occupancy state and how it relates to covariates can be considered constant within this timeframe (Appendix 3). These results indicate that a single analysis combining all data is reasonable. There was no evidence that detections among sites were dependent based on spatial autocorrelation of residuals of a logistic regression with the same

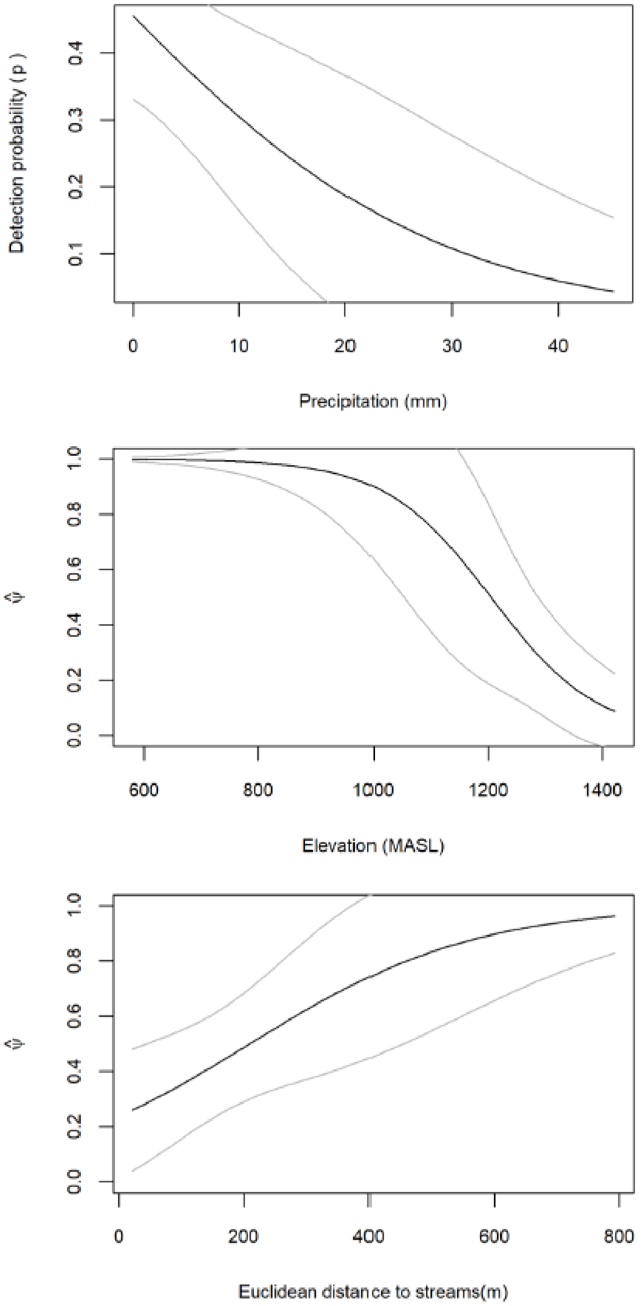


Figure 2. Estimated detection as a function of precipitation (in millimeters: mm) of the day previous to the survey (top), inferred occupancy as a function of elevation (middle) in meters above sea level (MASL) according to the best model for *Habia gutturalis*, and occupancy as a function of distance to streams according to the best model for *Capito hypoleucus* (bottom). Gray lines correspond to the 95% confidence intervals.

Table 3. Estimated coefficients and intercepts of occupancy and detection for the best models found for each species (all in logit scale). Standard errors (SE) for each estimated parameter are also provided. Model notation as explained in Table 1.

Species	Model	Parameters	Estimate	SE
White-mantled Barbet	$\psi(Dist.str)p(year)$	Year	-0.79	0.37
		Dist.str	0.96	0.51
		Ψ (intercept)	0.17	0.44
		P (intercept)	-0.48	0.32
Sooty Ant-Tanager	$\psi(Elev)p(year + Prec)$	Elevation	-2.82	1.31
		Year	-1.01	0.35
		Prec	-0.67	0.33
		Ψ (intercept)	1.65	1.27
		P (intercept)	-0.52	0.26

site covariates used in the occupancy models (Appendices 4 & 5). Nonetheless, we performed additional occupancy analyses excluding intermediate sites and we still identified the same best models as when using all data (Appendix 6). Therefore, we present next, results of analyses using all sites and surveys. The top model for the Sooty Ant-Tanager (*Habia gutturalis*) according to the Akaike information criterion corrected for small sample size included elevation as a covariate of occupancy and year and precipitation of the previous day as covariates of detection, whereas for the White-Mantled Barbet (*Capito hypoleucus*), the best model included distance to streams as a covariate of occupancy and year as a covariate of detection (Tables 1 & 2). The top model for the Sooty Ant-Tanager clearly stood out as the single best model accounting for almost 100% of the AIC weight and with all covariates of occupancy (elevation) and detection (year and precipitation of the previous day) showing coefficients whose 95% confidence intervals did not include 0 (Table 2, Fig. 2). The top model for the White-Mantled Barbet stood out from the other models proposed (Table 1). This model included distance to streams and year as covariate for occupancy and detection, respectively (Table 3). Occupancy of the Sooty Ant-Tanager decreased with elevation (Table 3, Fig. 2), while occupancy for the White-Mantled Barbet increased with distance to creeks. Our hypothesis that nest predation rates contributed to patterns of occupancy at these scales was not supported by the data (AIC weights of models that included predation were less than 5%, Tables

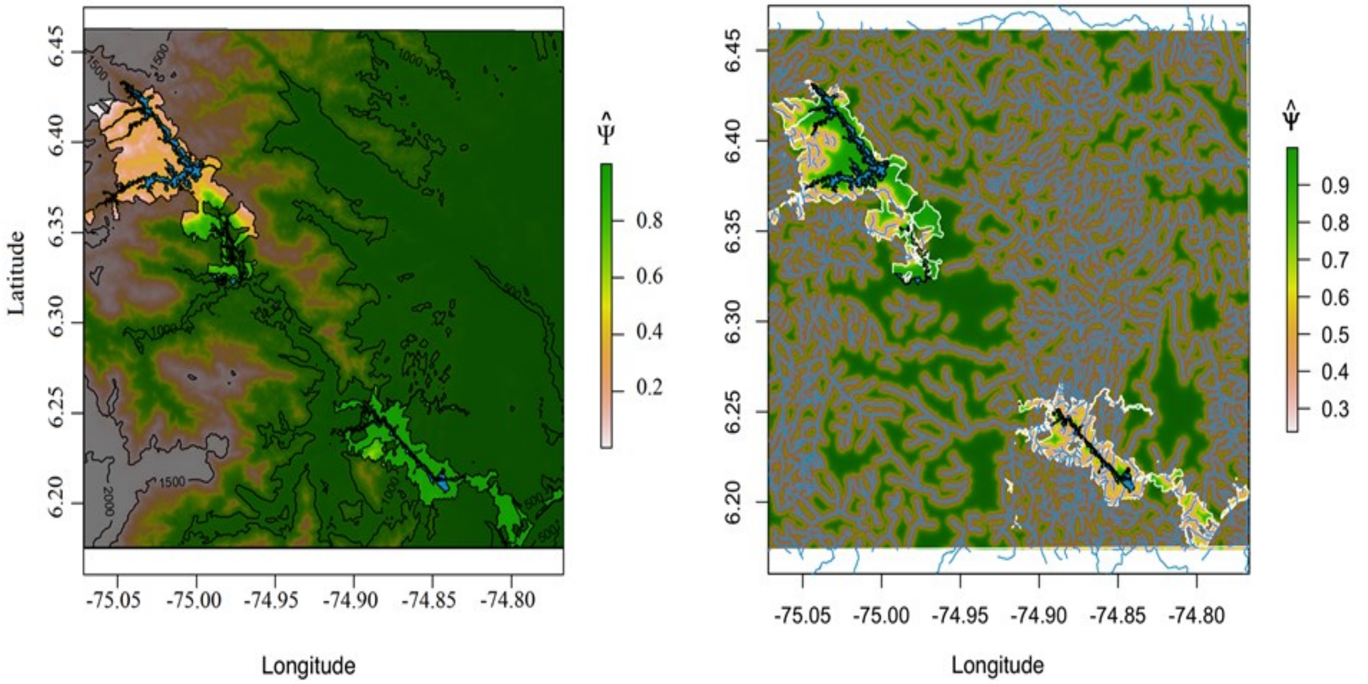


Figure 3. Occupancy probability in the buffer zones of each impoundment for *Habia gutturalis* (left panel) and *Capito hypoleucus* (right panel). Blue polygons represent Punchiná and San Lorenzo impoundments and highlighted areas represent the buffer zones of these two hydroelectric impoundments where the surveys took place. For *Habia gutturalis*, the map includes elevation isoclines every 500m intervals since this was the only covariate for occupancy in the best model for this species. For *Capito hypoleucus*, the map includes the streams (blue lines), since distance to streams was the only covariate for occupancy in the best model for this species.

1 and 2). According to the top models for both species, detection varied between years and was lower in 2015 (0.38±0.07 SE vs 0.21±0.05 for the White-Mantled Barbet, and 0.37±0.06 vs 0.17±0.04 for the Sooty Ant-Tanager, Table 3). For the Sooty Ant-Tanager, detection also decreased with increasing precipitation (Table 3).

The average occupancy in the buffer zones of each impoundment according to the best model for the Sooty Ant-Tanager (*H. gutturalis*) was higher in the buffer zone of the Punchiná impoundment (0.97 SD=0.053) than in San Lorenzo (0.39, SD=0.23), as expected given the relationship between occupancy and elevation (Fig. 3). On the other hand, average occupancy for the White-Mantled Barbet was higher in the buffer zone of the San Lorenzo impoundment (0.73, SD=0.25) than Punchiná (0.47, SD=0.17) and this was related to the relative lack of streams in the buffer zone of this impoundment (Fig. 3). We did not find evidence of overdispersion in our data for neither species (Appendix 7).

Discussion

In this study, we provide estimates of detection and occupancy for two endangered and endemic species in a fragmented landscape in the eastern flank of the Central Andes of Colombia, within the Magdalena Valley moist forest ecoregion. As far as we are concerned, these are the first estimates of occupancy for any threatened avian species in Colombia. Estimating occupancy is of practical and theoretical importance to define and reevaluate the conservation status of birds (Pearce & Ferrier 2001; Guarino *et al.* 2012, Kajzer *et al.* 2012). Our estimates indicate that occupancy varies according to elevation for the Sooty Ant-Tanager even along the narrow elevation range sampled (600-1500m). The Sooty Ant-Tanager has a higher occupancy in the buffer zones of the

Punchiná impoundment, which lies at a lower elevation than the San Lorenzo impoundment. On the other hand, distance to streams was the most informative covariate of occupancy for the White-Mantled Barbet. These results evidence that despite both species co-occur at regional extents, locally, they exhibit different relationships with landscape features. Results also indicate that the levels of deforestation encountered in the buffer zones of both impoundments are within the tolerance limits of both species since forest cover did not appear as an important covariate for neither species. This is consistent with the claim that the Sooty Ant-Tanager is resilient to some levels of deforestation (Willis 1972).

The Sooty Ant-Tanager was considered Near Threatened in the first edition of the Red Book of Birds of Colombia (Renjifo *et al*. 2002) but it was considered Least Concern according to the most recent edition (Renjifo *et al*. 2016). Our results indicate that the species is common within the study area and makes use of secondary forest and disturbed areas, more so when there is forest nearby, consistent with the results found by Laverde *et al*. (2005) in an area nearby on the opposite side of the Magdalena Valley. Our results are also consistent with those reported by Willis (1972), where he found an apparent increase in occupancy of this species in lightly disturbed environments. He cautioned though that this might only be a temporary situation and if the disturbance continued as is usual to the point where there is no vegetation cover along creeks, the populations may no longer persist (Willis 1972). It is important to notice that the landscapes found within the buffer zones of the two impoundments, might represent among the lowest disturbance levels in the region. On the other hand, our results indicate that occupation of the Sooty Ant-Tanager decreases sharply at the upper limit of its elevation range (1100m). Even though the mechanisms behind this limit are still unknown, it poses concerns about its potential ability to buffer changes due to climate change.

The White-Mantled Barbet is potentially less common than the Sooty Ant-Tanager, which is consistent with its current status as Endangered. Laverde *et al*. (2005) report that the species prefers forest with low intervention above 1000 m, and its abundance is related to the incidence of epiphytes, whose diversity is superior at intermediate elevations, mainly between 1000 and 2000 m where there is a greater relative humidity (Gentry & Dodson 1987). Our results indicate that variation in occupancy within the study area increases with the distance to streams. In the highly topographically heterogeneous area where these two species range, the crests of small hills are usually the farthest from streams, which are also the last areas to be deforested. This might be one of the reasons this species has been able to persist the high rates of land transformation.

Models that included vegetation-related or predation covariates had very little support based on AICw for neither the Sooty Ant-Tanager nor the White-Mantled Barbet (0.08), suggesting that deforestation levels in these buffer zones are still tolerable by both species and that nest predation might not be an issue to the viability of these species. Nest predation has been shown to be an important selective pressure on life history traits including nesting behavior, and one of the primary sources of mortality in tropical forests (Ricklefs 1969, Martin 1998, Ocampo & Londoño 2015). Recent studies indicate that predation pressure declines with increasing elevation, presumably as diversity and abundance of predators decrease (Jankowski *et al*. 2012, Roslin *et al*. 2017). Our measurements of nest predation did not show a relationship with elevation or with percentage forest cover (slopes not significantly different than 0 in ordinary least squares regression). It is also possible that the results of these experiments were not indicative of predation levels in cavities or above forest streams where the species of interest lay their nests (Willis 1961). The other site covariate that did not appear in the top models of neither species was distance to impoundments. Even though it is evident that proximity to water surface changes microclimate, this does not appear to affect occupancy levels.

Applying hierarchical models of occupancy, we estimated low detection probabilities for both species (range=0.17 – 0.38), despite the fact that they are medium-sized birds and highly but seasonally vocal. Most detections were auditory and at relatively low temperatures (~20°C), thus we recommend concentrating efforts on recording censuses, especially during reproductive peaks, and training personnel in song identification. The combination of acoustic monitoring (using autonomous species identification) and occupancy models can be a valuable tool to predict the distribution of a threatened species (Aide *et al.* 2013, Campos-Cerqueira & Aide 2016). In addition, the detection probability changed from year to year (~0.4 for 2014, ~0.2 for 2015 for both species). This could be partly based on a change of crew between years, but it could also be a consequence of a reduction in abundance due to increased deforestation and a higher frequency and intensity of dry periods (Blake & Loiselle 2016).

Occupancy models have gained recent popularity as one of the available methods that accounts for imperfect detection (Mackenzie *et al*. 2018). Nonetheless, the majority of ecological studies fail to correct for imperfect detection (Kellner & Swihart 2014). This could even be more drastic in the Neotropics, where various biodiversity hotspots converge, and where there is a special need for programs focused on biodiversity monitoring. We would like to encourage the use of this type of methods to monitor biodiversity, especially in cases where there is a strong suspicion that the methods employed for biodiversity detection are imperfect, which is definitely the case for birds.

According to the Red Book of Birds of Colombia (Renjifo *et al.* 2002, 2016), habitat loss and fragmentation are the most important threats for these two species. During the year 1987 and 2010, in the Magdalena valley moist forest region, where these two species are endemic to, the forested area has diminished in more than 60 % (Etter & Rodríguez 2008, Garzón & Gutiérrez 2013), and today it still represents a worrying core of deforestation (IDEAM 2017). Human activities are the main reason for landcover transformation. Within a conservation context, the buffer zones of hydroelectric impoundments can be conceived as reserves for some bird species despite their impacts in surrounding habitats including fragmentation. These protected areas give an opportunity for the conservation of endemic and threatened species such as the Sooty Ant-Tanager (*H. gutturalis*) and White-Mantled Barbet (*C. hypoleucus*), since they hold forested coverages with little human activity. Based on our results, we suggest that activities promoting connectivity among remaining forest patches are key to the preservation of these endemic birds. Continuous monitoring of these populations is necessary to evaluate not only spatial but temporal dynamics. We hope the data provided can be used in future evaluations of the conservation status of these birds (Spencer *et al.* 2011).

Acknowledgments

We like to thank the Grupo de investigación en Ecología y Evolución de Vertebrados and the Grupo Herpetológico de Antioquia (GHA) from Universidad de Antioquia for all the support and in particular Sergio Montoya, Henry Arenas, Jackeline Rivera, Hector Manuel Arango, Alvaro Rodriguez, Eliana Yepes, David Tobón, Sebastián Gallán, Juan Guillermo Mesa, Natalia Yepes, Maureen Daniela Montoya Ocampo and local personnel: Juan Carlos Quiceno and Olimpo Quintero for their assistance in the field. Lina Hinestroza and the personnel at each hydroelectric impoundment for their advice and support. Finally, we are grateful to David Ocampo for his support and advice in the nest predation experiment and to Alba Marin for her assistance in GIS analyses. We thank two reviewers for their comments and evaluation of the manuscript.

Funding statement.- All financial support was provided by ISAGEN, Universidad de Antioquia and Grupo Herpetológico de Antioquia (GHA), through Interinstitutional cooperation agreement 47/146, 2013-2015.

Ethics statement.- This research was conducted in compliance with the Guidelines to the Use of Wild Birds in Research.

Author contributions.- A.M.R. and J.L.P formulated the questions; A.M.R. and J.S.B. collected data; J.S.B, A.M.R, and J.L.P. analyzed the data; and A.M.R, J.S.B and J.L.P. wrote the paper.

Data deposits.- Our data is deposited with Dryad.

Literature cited

Aide, t. M., c. Corrada-bravo, m. Campos-cerqueira, c. Milan, g. Vega, & r. Alvarez. 2013. Real-time bioacoustics monitoring and automated species identification. Peerj 1:e103.

Anderson, e. P., c. N. Jenkins, s. Heilpern, j. A. Maldonado-ocampo, f. M. Carvajal-vallejos, a. C. Encalada, rivadeneira, juan f., hidalgo, max, cañas, carlos m., ortega, h., salcedo, norma, maldonado, mabel & tedesco, pablo a. 2018. Fragmentation of Andes-to-Amazon connectivity by hydropower dams. Science Advances 4:eaao1642.

Birdlife international. 2016a. *Capito hypoleucus* [www document]. IUCN Red List Threat. Species 2016. Url http://dx.doi.org/10.2305/iucn.uk.2016-3.rlts.t22681914a92925684.en.

Birdlife international. 2016b. *Habia gutturalis* [www document]. IUCN Red List Threat. Species 2016. Url http://dx.doi.org/10.2305/iucn.uk.2016-3.rlts.t22722423a94765906.en.

Blake, j. G., & b. A. Loiselle. 2015. Enigmatic declines in bird numbers in lowland forest of eastern Ecuador may be a consequence of climate change. Peerj 3:e1177.

Boyle, a. W., & b. J. Sigel. 2015. Ongoing changes in the avifauna of La Selva Biological Station, Costa Rica: twenty-three years of Christmas Bird Counts. Biological conservation 188:11–21.

Cardona, n., h. S. David, h. Gómez & p. Roldán. 2011. Flora de embalses, centrales hidroeléctricas de ISAGEN en el oriente antioqueño San Carlos, Jaguas y Calderas. Guía ilustrada. ISAGEN - Universidad de Antioquia, Herbario Universidad de Antioquia, Medellín, Colombia.

Campos-Cerqueira, M., & T. M. Aide. 2016. Improving distribution data of threatened species by combining acoustic monitoring and occupancy modelling. Methods in ecology and evolution 7:1340–1348.

Chandler, R. B., E. Muths, B. H. Sigafus, C. R. Schwalbe, C. J. Jarchow, & B. R. Hossack. 2015. Spatial occupancy models for predicting metapopulation dynamics and viability following reintroduction. Journal of Applied Ecology 52:1325–1333.

Diamond, J. M. 1972. Biogeographic kinetics: estimation of relaxation times for avifaunas of southwest pacific islands. Proceedings of the National Academy of Sciences 69:3199–3203.

Etter, A., & Rodríguez, N. 2008. Transformación de la cobertura forestal en el Valle del Magdalena Medio en el periodo de 1987-2001. Ambiente y Desarrollo, 12: 105-132.

Farnsworth, G. L., K. H. Pollock, J. D. Nichols, T. R. Simons, J. E. Hines, & J. R. Sauer. 2002. A removal model for estimating detection probabilities from point-count surveys. The AUK 119:414–425.

Feeley, K. J., & J. W. Terborgh. 2006. Habitat fragmentation and effects of herbivore (howler monkey) abundances on bird species richness. Ecology 87:144–150.

Finer, M., & C. N. Jenkis. 2012. Proliferation of hydroelectric dams in the Andean Amazon and implications for Andes-Amazon connectivity. PLoS ONE 4:e35126.

Fiske, I., & Chandler, R. 2011. Unmarked: an R package for fitting hierarchical models of wildlife occurrence and abundance. Journal of statistical software, 43: 1-23.

Garzón, N.V. & Gutiérrez, J. C. 2013. Deterioro de humedales en el Magdalena medio: un llamado para su conservación. Fundación Alma – Instituto de Investigación de Recursos Biológicos Alexander von Humboldt. Págs.:145.

Gentry, A. H., &. C. H. Dodson. 1987. Diversity and biogeography of neotropical vascular epiphytes. Annals of the Missouri Botanical Garden 74:205–233.

González, J., Etter, A., Sarmiento, A., Orrego, S., Ramirez, C., Cabrera, E., Vargas, D., Galindo, G., Garcia, M., & Ordoñez, m. 2011. Análisis de tendencias y patrones espaciales de deforestación en Colombia. Instituto de Hidrología, Meteorología y Estudios Ambientales-IDEAM. Bogotá D.C., Colombia. 64 p. Disponible en: http://www. Ideam.gov.co/documents/13257/13817/proyecciones.pdf/6cad956b-6b92-4320-a090-2000408a5765.

Guarino, E. D. S. G., A. M. Barbosa, & J. L. Waechter. 2012. Occurrence and abundance models of threatened plant species: applications to mitigate the impact of hydroelectric power dams. Ecological modelling 230:22–33.

Hilty, S.L., & W.L. Brown. 2001. Guía de las aves de Colombia. American Bird Conservancy, Imprelibros S. A., Cali, Colombia.

Holdridge, l. R. 1947. Determination of world plant formations from simple climatic data. Science 105:367–368.

Idárraga, P., Á., L.M. Urrea, F.J.P. Roldán & N.F.A. Cardona. 2016. Flora del Magdalena medio: áreas de influencia de la Central Térmica Termocentro. ISAGEN – Universidad de Antioquia, Herbario Universidad de Antioquia, Medellín (Colombia).

Isagen s.a. 2008. Centrales de generación isagen. Https://www.isagen.com.co/comunicados / centrales_generacion.pdf.

Jankowski, J. E., G. A. Londoño, S. K. Robinson, & M. A. Chappell. 2013. Exploring the role of physiology and biotic interactions in determining elevational ranges of tropical animals. Ecography 36:1–12.

Kajzer, J., M. Lenda, A. Kośmicki, R. Bobrek, T. Kowalczyk, R. Martyka, & P. Skórka. 2012. Patch occupancy and abundance of local populations in landscapes differing in degree of habitat fragmentation: a case study of the colonial black-headed gull, *Chroicocephalus ridibundus*. Journal of biogeography 39:371–381

Kellner, K. F., & Swihart, R. K. 2014. Accounting for imperfect detection in ecology: a quantitative review. Plos one 9: e111436.

Kéry, M., J. A. Royle, & H. Schmid. 2005. Modeling avian abundance from replicated counts using binomial mixture models. Ecological applications 15:1450–1461.

Kupferberg, S. J., W. J. Palen, A. J. Lind, S. Bobzien, A. Catenazzi, J. O. E. Drennan, & M. E. Power. 2012. Effects of flow regimes altered by dams on survival, population declines, and range-wide losses of California river-breeding frogs. Conservation biology 26:513–524.

Lahti, D. C. 2001. The edge effect on nest predation hypothesis after twenty years. Biological conservation 99:365–374.

Laverde, O., C. Munera, & L. M. Renjifo. 2005. Preferencia de hábitat por *Capito hypoleucus*, ave colombiana endémica y amenazada. Ornitología Colombiana 3:62–73.

Mackenzie, D. I., J. D. Nichols, G. B. Lachman, S. Droege, J. A. Royle, & C. A. Langtimm. 2002. Estimating site occupancy rates when detection probabilities are less than one. Ecology 83:2248–2255.

Mackenzie, D. I., & Bailey, L. 2004. Assessing the fit of site-occupancy models. Journal of agricultural, biological, and environmental statistics, 9:300-318.

Mackenzie, D. I., & J. T. Reardon. 2013. Occupancy methods for conservation management. In biodiversity monitoring and conservation: bridging the gap between global commitment and local action (B. Collen, N. Pettorelli, J. E. M. Baillie, and S. M. Durant, editors). Wiley-Blackwell, Oxford, UK.

Mackenzie, D. I., Nichols, J. D., royle, J. A., Pollock, K. H., Bailey, L., & Hines, J. E. 2018. Occupancy estimation and modeling: inferring patterns and dynamics of species occurrence. Academic press.

Martin, T. E. 1988. On the advantage of the being different: nest predation and the coexistence of bird species. Proceedings of the national academy of sciences 85:2196–2199.

Múnera, C. 2016. *Capito hypoleucus*. In libro rojo de aves de Colombia, volumen II: ecosistemas abiertos, secos, insulares, acuáticos continentales, marinos, tierras altas del Darién y Sierra Nevada de Santa Marta y bosques húmedos del centro, norte y oriente del país (L. M. Renjifo, A.M. Amaya-Villarreal, J. Burbano-Girón, & J. Velásquez-Tibatá, editors). Editorial Pontificia Universidad Javeriana e Instituto Alexander von Humboldt. Bogotá, Colombia.

Ocampo, D., & G. A. Londoño. 2015. Tropical montane birds have increased nesting success on small river islands. The Auk 132:1–10.

Pandit, M. K., & R. E. Grumbine. 2012. Potential effects of ongoing and proposed hydropower development on terrestrial biological diversity in the Indian Himalaya. Conservation biology 26:1061–1071.

Pearce, J., & S. Ferrier. 2001. The practical value of modelling relative abundance of species for regional conservation planning: a case study. Biological conservation 98:33–43.

Powell, L. L., N. J. Cordeiro, & J. A. Stratford. 2015. Ecology and conservation of avian insectivores of the rainforest understory: a pantropical perspective. Biological conservation 188:1–10.

Renjifo, L. M., A. M. Franco-Maya, J. M. Amaya-Espinel, G. H. Kattan, & B. López-Lanús. 2002. Libro rojo de aves de Colombia. Serie de libros rojos de especies amenazadas de Colombia. Instituto de investigación de recursos biológicos Alexander von Humboldt y ministerio de medio ambiente. Bogotá, Colombia.

Renjifo, L. M., M. F. Gómez, J. L. Velázquez, G. Kattan, J. D. Amaya, Á. M. Amaya & J. Burbano. 2014. Libro rojo de aves de Colombia. Volumen I: bosques húmedos de los andes y la costa pacífica. Bogotá, Colombia.

Renjifo, L. M., A. M. Amaya-Villarreal, J. Burbano-Girón & J. Velásquez-Tibatá. 2016. Libro rojo de aves de Colombia, volumen II: ecosistemas abiertos, secos, insulares, acuáticos continentales, marinos, tierras altas del Darién y Sierra Nevada de Santa Marta y bosques húmedos del centro, norte y oriente del país. Editorial Pontificia Universidad Javeriana e Instituto Alexander von Humboldt. Bogotá, colombia.

Ricklefs, R. E. 1969. The nesting cycle of songbirds in tropical and temperate regions. Living Bird 8:165–175.

Roslin, T., B. Hardwick, V. Novotny, W. K. Petry, N. R. Andrew, A. Asmus, I. C. Barrio, Y. Basset, A. L. Boesing, T. C. Bonebrake, & E. K. Cameron. 2017. Higher predation risk for insect prey at low latitudes and elevations. Science 356:742–744.

Royle, J. A., J. D. Nichols, & M. Kéry. 2005. Modelling occurrence and abundance of species when detection is imperfect. Oikos 110:353–359.

Simberloff, D. S., & E. O. Wilson. 1970. Experimental zoogeography of islands. A two-year record of colonization. Ecology 51:934–937.

Ideam, Sistema De Monitoreo De Bosques Y Carbono-SMBYC Instituto De Hidrología, Meteorología Y Estudios Ambientales. 2017. Décimo primer boletín de alertas tempranas de deforestación (at-d) segundo trimestre 2017.

Spencer, W., H. Rustigian-Romsos, J. Strittholt, R. Scheller, W. Zielinski, & R. Truex. 2011. Using occupancy and population models to assess habitat conservation opportunities for an isolated carnivore population. Biological conservation 144:788–803.

Stiles, F. G., L. Rosselli, & C. I. Bohórquez. 1999. New and noteworthy records of birds from the middle Magdalena valley of Colombia. Bulletin-British Ornithologists Club 119:113–128.

Stratford, J. A., & P. C. Stouffer. 1999. Local extinctions of terrestrial insectivorous birds in a fragmented landscape near Manaus, Brazil. Conservation biology 13:1416–1423.

Terborgh, J., L. Lopez, P. Nuñez, M. Rao, G. Shahabuddin, G. Orihuela, M. Riveros, R. Ascanio, G. H. Adler, T. D. Lambert, & L. Balbas. 2001. Ecological meltdown in predator-free forest fragments. Science 294:1923–1926.

Thiollay, J. M. 1997. Disturbance, selective logging and bird diversity: a neotropical forest study. Biodiversity and conservation 6:1155–1173.

Wilcove, D. S. 1985. Nest predation in forest tracts and the decline of migratory songbirds. Ecology 66:1211–1214.

WCS, Wildlife Conservation Society. 2019. Magdalena [www document]. Wcs colomb. Url https://colombia.wcs.org/es-es/paisajes/andes/magdalena.aspx.

Willis, E. 1961. A study of nesting ant-tanagers in British Honduras. The condor 63:479-503.

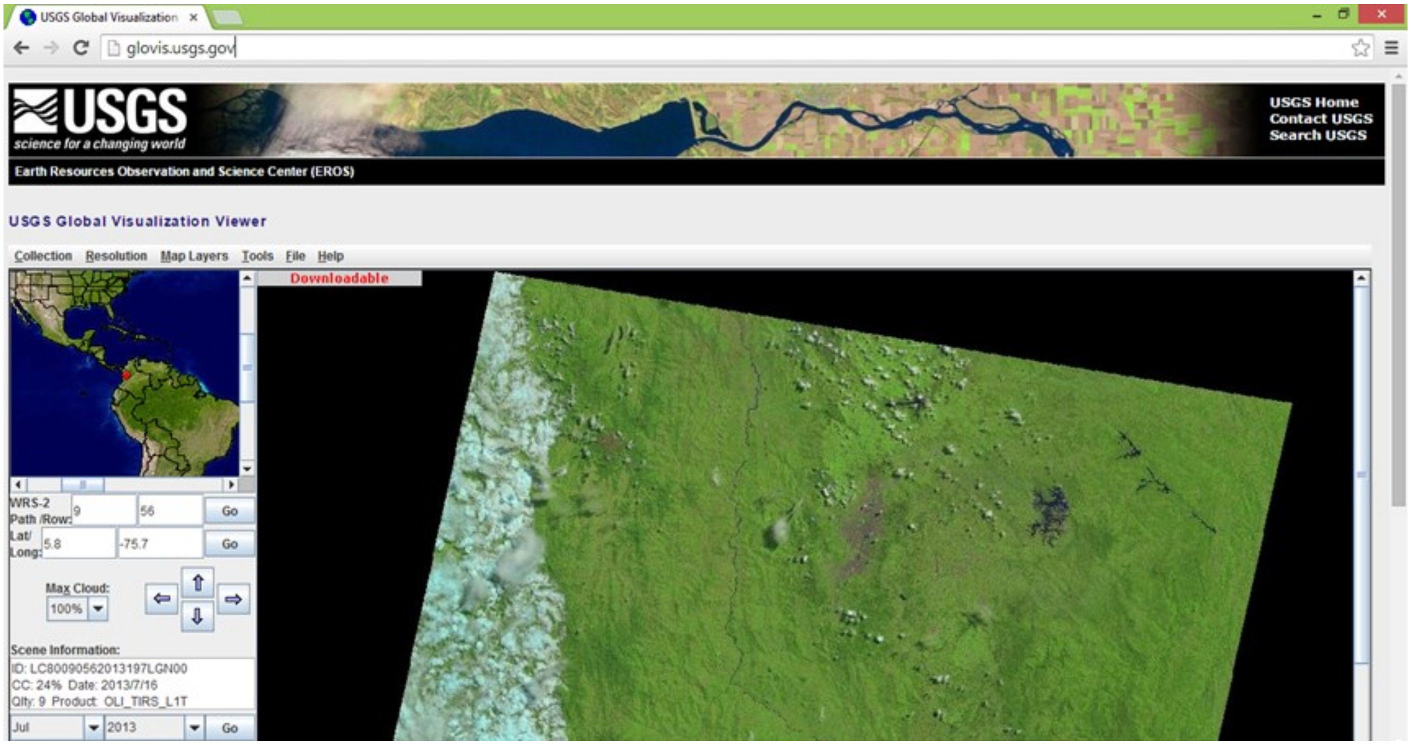
Willis, E. O. 1972. Taxonomy, ecology, and behavior of the sooty ant-tanager (*Habia gutturalis*) and other ant-tanagers (aves). American Museum Novitates no. 2480.

Ziv, G., E. Baran, A. Nam, I. Rodríguez-Iturbe, & S. A. Levin. 2012. Trading-off fish biodiversity, food security, and hydropower in the mekong river basin. Proceedings of The National Academy of Sciences 109:5609–5614.

Recibido: 18 de noviembre de 2018 *Aceptado:* 24 de agosto de 2020

Citación: Betancur, J.S., A. Morales-Rozo & J.L. Parra. 2020. Occupancy of the Sooty ant-tanager (*Habia gutturalis*) and White-mantled barbet (*Capito hypoleucus*) in fragmented forests of the Central Andes in Colombia. Ornitología Colombiana 18:eA03.

Appendix 1. **Vegetation cover.** In order to generate a landcover surface for the study area, we used a Landsat TM 8 satellite image from July 2013 (http://glovis.usgs.gov/) with a 24% cloud cover for the entire image but no cover for our study area (Fig. SM1). We used a supervised classification analysis in ArcGis 10.0 with 35 GPS control points with known vegetation or cover type determined on ground. The CORINE Land Cover method is the official land cover classification used in Colombia (IDEAM 2010). This method consists of a training phase, where features in the image that allow to characterize the six main cover types (pastures, mosaics, fragmented forest, water bodies, bare or degraded land, and urban areas) are identified. Based on the spectral signature of each cover type defined in the training phase, pixels are then categorized using a maximum likelihood approach into each possible category. Finally, the classifier is evaluated using the control points with known vegetation cover. The result of this classification is a vegetation cover raster with 1-hectare resolution (Fig. SM2).



WRS-2 Path /Row: 9 56

Lat/ Long: 5.8 -75.7

Max Cloud: 100%

Scene Information:
ID: LC80090562013197LGN00
CC: 24% Date: 2013/7/16
Qlty: 9 Product: OLI_TIRS_L1T

Jul 2013

Figure SM1. Landsat TM satellite image used. Red square highlights the study area.

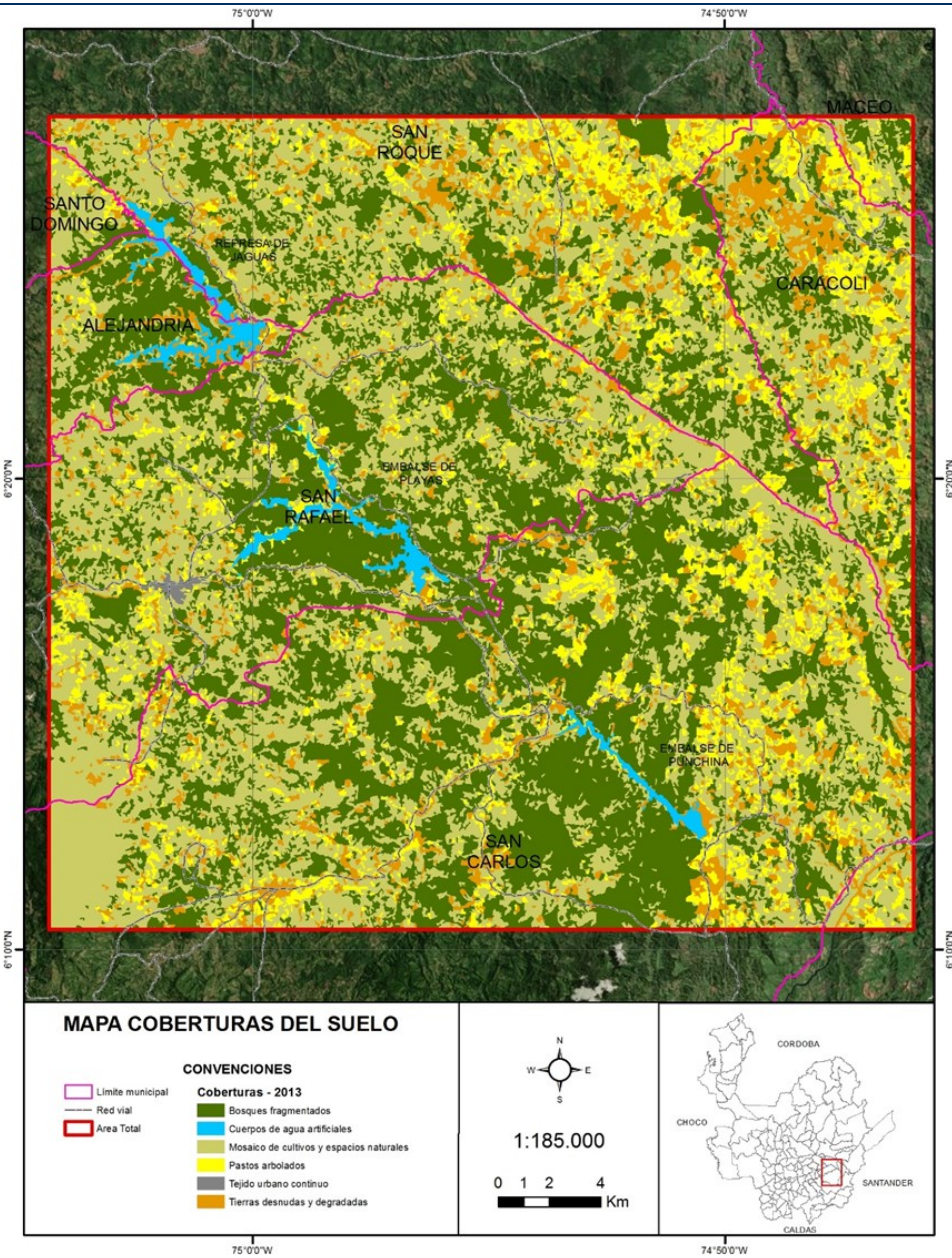


Figure SM2. Land cover type raster for the study area.

Literature cited

IDEAM. 2010. Leyenda Nacional de Coberturas de la Tierra. Metodología CORINE Land Cover adaptada para Colombia Escala 1:100.000. Instituto de Hidrología, Meteorología y Estudios Ambientales. Bogotá, D. C., 72p.

Appendix 2. Nest predation experiments.

Figure SM3. Artificial cup nests employed in the predation experiment. Each nest was exposed to heights between 0.5 and 1.5 m not less than 10 m distance among them within every sampling unit (point count). The nest contained one common quail egg (*Coturnix coturnix*) and one imitation plasticine egg.

Appendix 3. **Occupancy models grouping the surveys by year.** Occupancy models were generated for each year to corroborate whether occupancy state and its relations with covariates for each species were similar between the first (June 2014) and second group of surveys (July 2015).

Occupancy models for *Habia gutturalis*

Table SM1. Results from single season occupancy models evaluated for each year for *Habia gutturalis*. The models are presented in descending order based on AICc values. The best model identified for both years also includes elevation as the most informative covariate. Points within parentheses indicate models without covariates. The five models where occupancy is constant (.) were evaluated first to identify the most likely covariates for detection. The shortest straight distance to streams (Dist.str), the shortest straight distance to impoundments (Dist. imp), time of the survey (H.max), the average temperature during the survey (Temp), and the precipitation of the previous day (Prec).

Year	Model	Number of parameters	AICc	ΔAICc	AIC$_w$
2014	$\boldsymbol{\psi(Elevation)\rho(.)}$	3	142.18	0.00	≈1.00
	$\psi(Mosaic)\rho(.)$	3	155.81	13.63	<0.001
	$\psi(Dist.str.)\rho(.)$	2	156.68	14.49	<0.001
	$\psi(.)\rho(.)$	3	158.40	16.22	<0.001
	$\psi(.)\rho(prec)$	3	158.50	16.31	<0.001
	$\psi(Forest)\rho(.)$	3	158.51	16.33	<0.001
	$\psi(Dist.imp)\rho$	3	158.55	16.36	<0.001
	$\psi(.)\rho(Temp)$	3	158.61	16.43	<0.001
	$\psi(.)\rho(H.max)$	3	158.64	16.45	<0.001
2015	$\boldsymbol{\psi(Elevation)\rho}(.)$	3	109.73	0.00	0.74
	$\psi(Mosaic)\rho(.)$	3	114.35	4.97	0.06
	$\psi(Dist.str.)\rho(.)$	3	114.95	5.58	0.04
	$\psi(.)\rho(.)$	2	115.19	5.82	0.04
	$\psi(.)\rho(prec)$	3	115.39	6.01	0.04
	$\psi(Forest)\rho(.)$	3	116.05	6.67	0.03
	$\psi(Dist.imp)\rho$	3	116.56	7.19	0.02
	$\psi(.)\rho(Temp)$	3	117.00	7.63	0.02
	$\psi(.)\rho(H.max)$	3	117.14	7.76	0.01

Table SM2. Estimated coefficients and intercepts of occupancy and detection for the best models found for *Habia gutturalis* (all in logit scale). Standard errors (SE) for each estimated parameter are also provided. Values obtained for the best models are similar between years.

Year	Model	Parameters	Estimate	SE
2014	$\psi(Elev)p(.)$	Elevation (Elev)	-3.15	1.63
		Ψ (intercept)	1.67	1.56
		P (intercept)	-0.36	0.25
2015	$\psi(Elev)p(.)$	Elevation (Elev)	-2.70	1.98
		Ψ (intercept)	1.66	2.16
		P (intercept)	-1.34	0.32

Occupancy models for *Capito hypoleucus*

Table SM3. Results from single season occupancy models evaluated for each year for *Capito hypoleucus*. The models are presented in descending order based on AIC values. The best model for 2014 is consistent with the one that includes the distance to the streams as a covariate. For the year 2015, instead, it includes the distance to the dam as the most important covariate, however, the second-best model includes the distance to the streams as the best covariate and the AIC value is not considerably different between models (ΔAICc<2). Points within parentheses indicate models without covariates. The five models where occupancy is constant (.) were evaluated first to identify the most likely covariates for detection. The shortest straight distance to streams (Dist.str), the shortest straight distance to impoundments (Dist. imp), time of the survey (H.max), the average temperature during the survey (Temp), and the precipitation of the previous day (Prec).

Year	Model	Number of parameters	AICc	ΔAICc	AIC_w
2014	$\psi(Dist.str)\rho(.)$	**3**	**127.95**	**0**	**0.88**
	$\Psi(Elev)\rho(.)$	3	134.38	6.43	0.03
	$\Psi(.)\rho(.)$	2	135.66	7.71	0.02
	$\Psi(Dist.imp)\rho(.)$	3	136.54	8.59	0.01
	$\Psi(.)\rho(Temp)$	3	136.66	8.71	0.01
	$\Psi(Forest)\rho(.)$	3	137.01	9.06	0.01
	$\Psi(Mosaic)\rho(.)$	3	137.04	9.09	0.01
	$\Psi(.)\rho(H.max)$	3	137.11	9.16	0.01
	$\Psi(.)\rho(prec)$	3	137.29	9.34	0.01
2015	$\Psi(Dist.imp)\rho(prec)$	**4**	**99.18**	**0**	**0.42**
	$\psi(Dist.str)p(prec)$	**4**	**100.77**	**1.59**	**0.19**
	$\Psi(.)\rho(prec)$	3	101.99	2.82	0.10
	$\Psi(Elev)\rho(prec)$	4	102.74	3.57	0.07
	$\Psi(.)\rho(H.max)$	3	102.95	3.77	0.06
	$\Psi(.)\rho(.)$	2	103.5	4.32	0.05
	$\Psi(Forest)\rho(prec)$	4	103.78	4.61	0.04
	$\Psi(Mosaic)\rho(prec)$	4	103.97	4.79	0.04
	$\Psi(.)\rho(Temp)$	3	105.49	6.31	0.02

Table SM4. Estimated coefficients and intercepts of occupancy and detection for the best models found for *Capito hypoleucus* (all in logit scale). Standard errors (SE) for each estimated parameter are also provided. Parameter values between models are not similar. However, the relationship between the covariate distance to the streams and the occupancy of the species remains positive between years. Models from 2015 didn't converge.

Year	Model	Parameters	Estimate	SE
2014	$\psi(Dist.str)p(.)$	Dist.str	1.22	0.54
		Ψ (intercept)	-0.40	0.47
		P (intercept)	-0.04	0.43
2015	$\Psi(Dist.imp)\rho(Prec)$	Dist.imp	18.70	20.4
		Prec	-0.58	0.41
		Ψ (intercept)	11.90	13.90
		P (intercept)	-1.76	0.31
2015	$\psi(Dist.str)p(Prec)$	Dist.str	33.00	43.90
		Prec	-0.63	0.41
		Ψ (intercept)	34.80	43.80
		P (intercept)	-1.95	0.29

Appendix 4. Logistic regression. In order to assess congruency of our results with other techniques, we used overall detection/non-detection results (Detection) for each site and modeled them with a logistic regression in R using the glm function with the binomial link with each of the occupancy covariates (distance to streams, percent forest cover, percent mosaic cover, elevation, distance to impoundment, and nest predation). We identified the best model using AIC and we used the residuals of the best model in a spatial correlogram to evaluate if sites were spatially dependent (see Appendix 5).

Table SM5. Results of logistic regression models for *Capito hypoleucus* presented in decreasing order according to their AIC value. The best model according to AICc was the same best model identified with the occupancy analysis: the one including the shortest straight distance to streams (Dist.str) as a covariate. The five models where occupancy is constant (.) were evaluated first to identify the most likely covariates for detection. The shortest straight distance to streams (Dist.str), the shortest straight distance to impoundments (Dist. imp), time of the survey (H.max), the average temperature during the survey (Temp), the precipitation of the previous day (Prec).

Model	df	AICc
Detection ~ Dist.str	2	92.85
Detection ~ 1	1	96.28
Detection ~ Forest	2	96.44
Detection ~ Elevation	2	96.46
Detection ~ Mosaic	2	97.35
Detection ~ Dist.imp	2	98.25
Detection ~ Depredation	2	98.33

Table SM6. Estimated parameters (intercept and coefficient of covariate) of the best model found for *Capito hypoleucus*. The sign in parentheses represents the direction of the relationship.

Model	Coefficients	Estimate	SE
Detection ~ *Dist.str*	Intercept	-1.26	0.46
	Distance stream (Dist. str)	0.00	0.00

Table SM7. Results of logistic regression models for *Habia gutturalis* presented in decreasing order according to their AIC value. The best model according to AICc with logistic regression was the same best model identified with the occupancy analysis: the one including elevation as a covariate.

Model	df	AICc
Detection ~ Elevation	2	84.97
Detection ~ Dist.imp	2	97.85
Detection ~ 1	1	98.87
Detection ~ Forest	2	99.25
Detection ~ Depredation	2	99.48
Anwer ~ Dist.str	2	100.70
Anwer ~ Mosaic	2	100.97

Table SM8. Estimated parameters (intercept and coefficient of covariate) of the best model found for *Habia gutturalis*. The sign in parentheses represents the direction of the relationship.

Model	Coefficients	Estimate	SE
Answer ~ Elevation	Intercept	1.81	1.06
	Elevation	-0.004	0.00

Appendix 5. Spatial autocorrelation (Moran's I) of residuals of logistic regression. In order to quantitatively assess whether there was spatial dependence in our observations after accounting for the effect of our covariates, we performed a spatial autocorrelation (Moran's I) of the residuals of the logistic regression (see Appendix 3). This test evaluates whether the residuals of point counts that are closer in space are more similar. We used the dnearneigh function in conjunction with the sp.cor function from the spdep package in R to perform these analyses.

For *Capito hypoleucus* and *Habia gutturalis* no significant (P<0.05) spatial autocorrelation was found for residuals at any of the three closest distance classes between 0.1 and 1 km (Tables SM8; Fig. SM5). Correlation of residuals was higher at the shortest distance class (lag 1, Moran's I~0.18) for both species and thus, an additional occupancy analysis was performed by leaving out intermediate points (see Appendix 5).

Table SM9. Results of spatial autocorrelation analyses (Moran's I) of residuals from a logistic regression between Detection/Non-detection and distance to streams for *Capito hypoleucus*. Lag represents the distance classes and n is the number of comparisons within each distance class.

Lag (n)	Moran's I	Expectation	Variance	Standard Deviate	Pr(I) two sided
1 (70)	0.19	-0.01	0.01	23.53	0.06
2 (30)	0.12	-0.03	0.04	0.82	100.00
3 (16)	0.18	-0.07	0.02	15.54	0.36

Table SM10. Results of spatial autocorrelation analyses (Moran's I) of residuals from a logistic regression between Detection/Non-detection and elevation for *Habia gutturalis*. Lag represents the distance classes and n is the number of comparisons within each distance class. The asterisk indicates instances where the probability of the null hypothesis of no autocorrelation is <0.05.

Lag (n)	Moran's I	Expectation	Variance	Standard Deviate	Pr(I) two sided
1 -70	0.17	-0.01	0.01	21.94	0.08
2 -30	0.19	-0.03	0.04	11.69	0.73
3 -16	0.16	-0.07	0.02	13.88	0.49

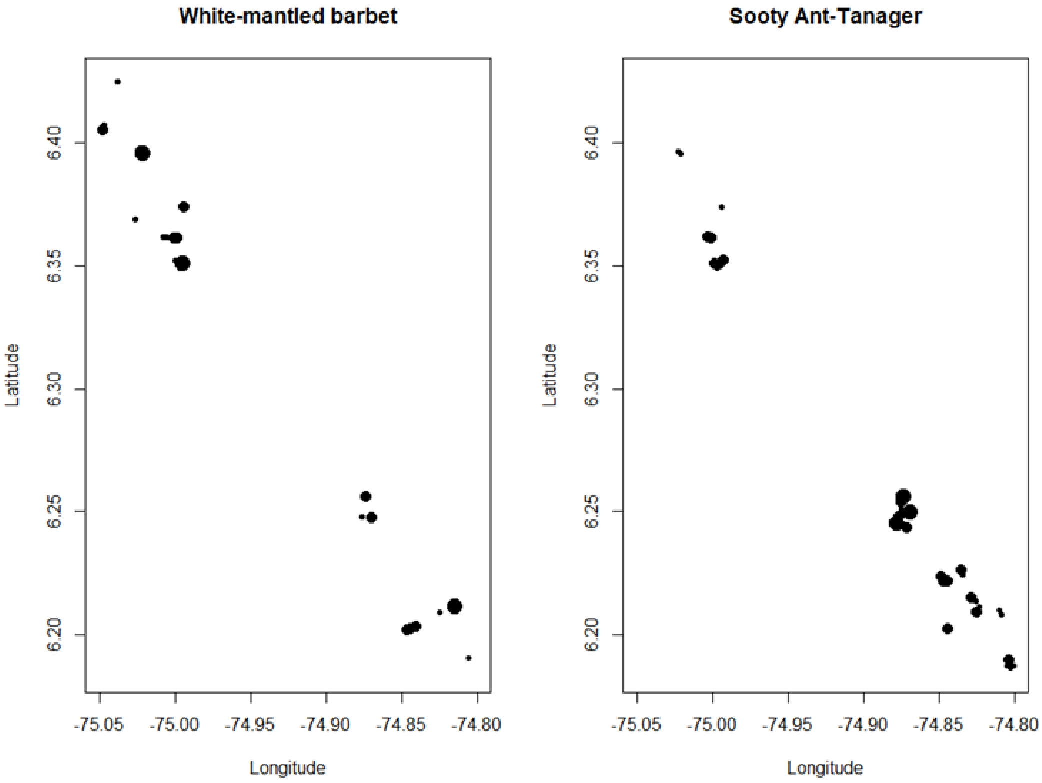


Figure SM4. Distribution of point counts in the buffer zones of both impoundments for each species showing all the point counts and if distances between 0.1 and 1km connect them. Size of the dot represents the amount of detections at each site.

Appendix 6. Occupancy models without intermediate point counts. Occupancy models were generated without intermediate point counts for each species.

Table SM11. Results from single season occupancy models evaluated without intermediate points for *Capito hypoleucus*. The models are presented in descending order based on AICc values. Even when eliminating intermediate point counts, the best model identified also includes euclidean distance to streams as the most informative covariate. However, year is no longer identified as an important covariate for detectability. Points within parentheses indicate models without covariates. The five models where occupancy is constant (.) were evaluated first to identify the most likely covariates for detection. The shortest straight distance to streams (Dist.str), the shortest straight distance to impoundments (Dist. imp), time of the survey (H.max), the average temperature during the survey (Temp), and the precipitation of the previous day (Prec).

Model	Number of parameters	AICc	ΔAICc	AIC_w
$\psi(Dist.str.)\rho(.)$	3	109.95	0	0.65
$\psi(.)\rho(.)$	2	114.36	4.41	0.06
$\psi(.)\rho(H.max)$	3	114.90	4.95	0.05
$(Elevation)\rho(.)$	3	115.56	5.61	0.04
$\psi(.)\rho(Prec)$	3	115.62	5.67	0.04
$\psi(.)\rho(Temp)$	3	115.96	6.01	0.03
$\psi(Forest)\rho(.)$	3	116.41	6.46	0.03
$\psi(Predation)\rho(.)$	3	116.60	6.65	0.03
$\psi(.)\rho(year)$	3	116.48	6.53	0.02
$\psi(Dist.imp)\rho.$	3	116.73	6.78	0.02
$\psi(Mosaic)\rho(.)$	3	116.74	6.79	0.02

Table SM12. Results of single season occupancy models evaluated without intermediate points for *Habia gutturalis*. The models are presented in descending order based on AICc values. Even when eliminating intermediate point counts, the best model identified also includes elevation as the most informative covariate. However, precipitation is no longer identified as an explanatory covariate for detectability. Points within parentheses indicate models without covariates. The five models where occupancy is constant (.) were evaluated first to identify the most likely covariates for detection. The shortest straight distance to streams (Dist.str), the shortest straight distance to impoundments (Dist. imp), time of the survey (H.max), the average temperature during the survey (Temp), and the precipitation of the previous day (Prec).

Model	Number of parameters	AICc	ΔAIC	AIC_w
$\psi(Elevation)\rho(year + H.max)$	5	123.27	0	≈1
$\psi(Dist.imp)\rho(year + H.max)$	5	128.80	5.53	<0.001
$\psi(.)\rho(year + +H.max)$	4	131.25	7.98	<0.001
$\psi(Mosaic)\rho(year + +H.max)$	5	132.08	8.81	<0.001
$\psi(Forest)\rho(year + H.max)$	5	133.85	10.58	<0.001
ψ(Predation)ρ(year+H.max)	5	133.78	10.51	<0.001
$\psi(Dist.str.)\rho(year + H.max)$	5	133.97	10.7	<0.001
$\psi(.)\rho(H.max)$	3	135.99	12.72	<0.001
$\psi(.)\rho(year)$	3	136.30	13.03	<0.001
$\psi(.)\rho(.)$	2	141.81	18.54	<0.001
$\psi(.)\rho(Prec)$	3	142.23	18.96	<0.001
$\psi(.)\rho(Temp)$	3	144.12	20.85	<0.001

Appendix 7. Overdispersion. We performed a goodness-of-fit test for single season occupancy models to evaluate for overdispersion in our best models for *Habia gutturalis* and *Capito hypoleucus*. These functions compute the MacKenzie and Bailey (2004) goodness-of-fit test for single season occupancy models based on Pearson's chi-square (Mazerolle & Mazerolle 2019).

Overdispersion test for *Capito hypoleucus* (ψ (Dist.str) ρ(year)) According to the results of this test, there is no evidence of overdispersion in our data. The p-value is greater than 0.05 and the chi-square is not large (Fig SM4). We can be confident that the best model for *Capito hypoleucus* has an adequate fit to the data captured.

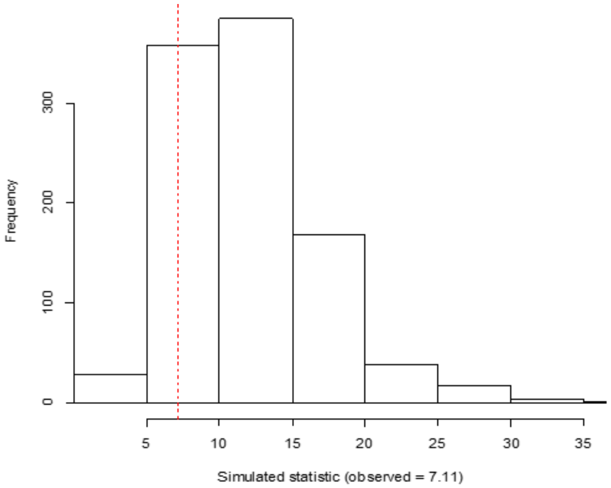


Figure SM5. Bootstrapped MacKenie and Bailey fit statistic (1000 samples) for *Capito hypoleucus*. P-value= 0.877. Chi-square statistic = 7.11. Estimate of c-hat= 0.59.

Overdispersion test for *Habia gutturalis* (ψ (Dist.str) ρ(year)) According to the results of this test, there is no evidence for overdispersion in our data. The p-value is greater than 0.05 and the chi-square is not very large. The p-value is greater than 0.05 and the chi-square is not very large (Fig. SM5). We can be confident that the best model for *Habia gutturalis* has an adequate fit to the data captured.

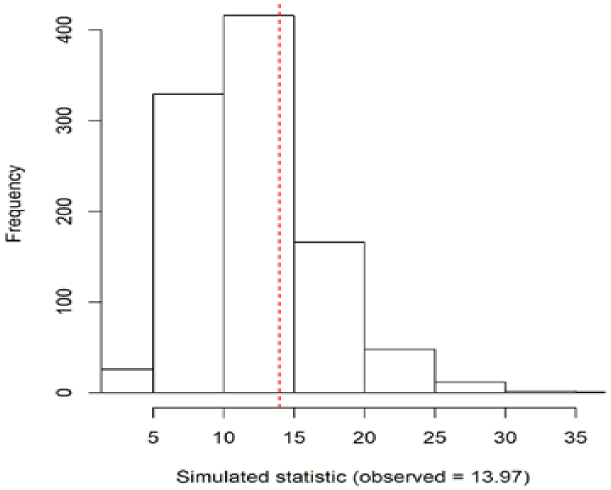


Figure SM6. Bootstrapped MacKenzie and Bailey fit statistic (1000 samples) for *Habia gutturalis*. P-value= 0.371. Chi-square statistic = 13.97. Estimate of c-hat= 1.08.

Literature cited

MacKenzie, D. I., & Bailey, L. L. 2004. Assessing the fit of site-occupancy models. Journal of Agricultural, Biological, and Environmental Statistics, 9(3): 300-318.

Mazerolle, M. J., & Mazerolle, M. M. J. 2019. Package 'AICcmodavg'.