

Occupancy Modeling for the Conservation Assessment of the Peruvian Night Monkey (*Aotus miconax*)

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Abstract: The Peruvian night monkey, *Aotus miconax*, currently listed by IUCN as Endangered, is endemic to the cloud forests of northeastern Peru, in the Tropical Andean Biodiversity Hotspot. Its distribution, abundance and habitat use remain understudied throughout its range. This study involved detecting and comparing the presence or absence of *A. miconax* in two degraded forests using occupancy modeling, a method that has proven to be more appropriate for steep terrain and dense vegetation compared to line transects. The two study areas were stratified and partitioned into 40 survey points. Each point was visited five times for 20 minutes to determine by visual, auditory or other species-specific signs, whether each point was ‘occupied’ by the target species. We analyzed the data with PRESENCE to infer species abundance and detectability. We detected *A. miconax* at 13 sites out of 40, producing a naïve probability of occupancy of 0.33. Altitude, canopy cover, and tree diversity were important habitat covariates affecting occupancy; the best model shows a probability of occupancy of 0.51 for canopy cover and tree diversity. Moon phase and tree diversity were important in detectability. Detection/non-detection data proved to be an efficient approach for elucidating the spatial distribution and habitat requirements of *A. miconax*. The results support the use of occupancy modeling as a means of surveying arboreal primate species in difficult terrains and habitats, especially for large areas in a limited time, and should be further tested for application with other species and ecological niches.

Key words: Amazonas, PRESENCE, habitat, abundance, Methods, Montane forest

Introduction

Monitoring wildlife populations for changes in abundance and distribution is an important element in decision making for biodiversity conservation (Nichols *et al.* 2008; Guillera-Aroita *et al.* 2010). Monitoring programs are essential considering the current lack of information on many species and the continued loss of biodiversity (Buckland *et al.* 2006; Nekaris *et al.* 2007). Monitoring processes need to be not only sustainable and consistent but also straightforward and cost effective (MacKenzie *et al.* 2006). Most importantly, long-term monitoring methods must allow accurate inferences in changes in population dynamics (Martin 2003; Strier and Mendes 2009; Buckland *et al.* 2010).

Among the variety of techniques employed in monitoring programs, line transects are the most widely applied to determine primate abundance and density, including for nocturnal primates: galagos (Weisenseel *et al.* 1993; Bearder *et al.* 2003); lorises (Nekaris *et al.* 2005, 2007); lemurs (Ganzhorn

2002); and night monkeys (Wright 1989; Aquino and Encarnación 1994; Fernandez-Duque *et al.* 2010). Line transects, however, have been shown to be less suitable when survey areas are mountainous (Li and Rogers 2007) or if the vegetation is particularly dense, and it has been argued that nocturnal animal populations might be underestimated (Duckworth 1998). Besides, the need for access may also influence the location of line transects, which will bias the data collected (Bibby *et al.* 1992; Ross and Reeve 2003). Fixed-point sampling, which has been widely used in songbird (Bibby *et al.* 1992; Kissling and Garton 2006), small mammal (Odel and Knight 2001; Ruethe *et al.* 2003), and fish abundance surveys (Copp 2010), can be applied as an alternative survey method to line transects.

Points are easier to locate randomly without the restriction of dense vegetation and are also more suited to mountainous terrain (Green 1978; Hanya *et al.* 2003). The use of auditory sampling at fixed points has been developed to estimate primate density when the target species occupies the upper

strata of the forest and/or produce loud vocalizations, such as indri and certain colobines and gibbons (Nijman 2001; Nijman and Menken 2005; Buckley *et al.* 2006). To date, few nocturnal primate surveys have incorporated the point-sampling method for monitoring purposes, even though many nocturnal species, including the galagos (Bearder 2007), slow lorises (Zimmermann 1985), mouse lemurs (Braune *et al.* 2005), night monkeys (Wright 1989) and tarsiers (Nietsch and Kopp 1998) vocalize often and should lend themselves to such a method.

Occupancy modelling is a species population assessment technique that estimates the proportion of an area that is occupied by a species through modelling detection/non-detection (Bailey and Adams 2005). Used in conjunction with fixed point sampling, it is often sufficient for monitoring objectives, considering it is less time demanding and is suitable for rapid assessments if financial resources are limited (MacKenzie *et al.* 2002, 2006). Occupancy modelling can also be used to assess habitat preferences for a species, as well as sources of anthropogenic disturbance (MacKenzie 2005; Olson *et al.* 2005). Although frequently used in bird, amphibian, and reptile studies (Poulsen and Krabbe 1998; Kéry 2002), occupancy modelling is still a novel concept in primate research (Guillera-Arroita *et al.* 2010; Karanth *et al.* 2010).

The main purpose of this survey was to test whether fixed-point sampling with occupancy modelling worked as a census technique for surveys of the nocturnal Peruvian night monkey *Aotus miconax*, a poorly studied primate endemic to the cloud forests of Peru. Occupancy and detectability were modelled as functions of habitat characteristics (elevation, diversity of trees at the genus level, and canopy cover) to determine if they were associated with the presence of *A. miconax*. If affected by anthropogenic habitat changes, then we would expect *A. miconax* to occupy sites characterized by greater vegetation diversity and canopy continuity. We also predicted higher detectability of the species on brighter nights, as peak nocturnal activity of night monkeys is associated with the full moon (Fernandez-Duque *et al.* 2010) and because of better visibility.

Methods

The Peruvian night monkey

The Peruvian night monkey *Aotus miconax* is characterized by its red neck. It occurs in the departments of Amazonas and San Martín (Fig. 1), in primary and secondary montane forests on steep slopes to the timberline and remnant forests at elevations between 1,200 and 3,100 m above sea level (Butchart *et al.* 1995; Marchena *et al.* 2011; Shanee *et al.* 2015). Due to habitat loss and its restricted range, *A. miconax* is listed by IUCN and the Peruvian Government as Endangered, with a declining population (El Peruano, Decreto Supremo 34-2004-AG, 22 September 2004; Cornejo *et al.* 2019; Shanee *et al.* 2015). The few published studies of *A. miconax* cover basic natural history, distribution, conservation

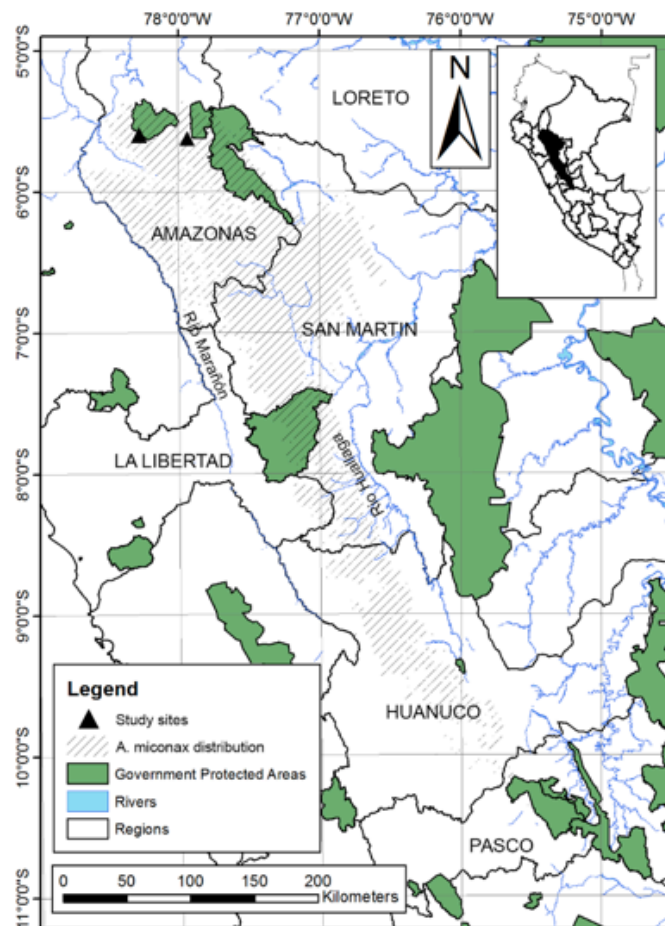


Figure 1. Map indicating estimated distribution of *Aotus miconax*, with location of study area in the department of Amazonas highlighted.

status and threats (Butchart *et al.* 1995; Cornejo *et al.* 2008; Shanee and Shanee 2011; Shanee *et al.* 2013, 2015).

Survey sites

During the survey period (the dry season between May and July 2011), we sampled along 6.1 km of trails in two forested areas: Cabeza del Toro and the Santuario Nacional Cordillera de Colán, both in the Amazonas department of northeastern Peru. Cabeza del Toro (5°38'S, 77°54'W) is a community-owned forest in the 82,000-ha Comunidad Campesina Yambrasbamba. It is situated approximately 2,000 m. above sea level in an area bordered by the Zona Reservada Río Nieva, Área de Conservación Privada Abra Patricia, Santuario Nacional Cordillera de Colán and Área de Conservación Privada Pampa del Burro. Cabeza del Toro is a fragmented mosaic of disturbed primary and secondary cloud forest of about 300 ha; interspersed with cattle pasture (Shanee and Shanee 2015). The area is not legally protected.

The Santuario Nacional Cordillera de Colán (5°32.52'S, 78°11.29'W) was created in 2009. It protects an area of about 39,000 ha of high-altitude montane cloud forest (Peru, Ministerio del Ambiente, 2009; Navarro 2010). The survey site was located in the 5-km buffer zone that surrounds the protected area. Logging of cedar wood is common in this area of

the buffer zone, which was established to protect the remaining forest. It was evident during our surveys, however, that the national sanctuary area was being logged as well. Cattle ranching was prevalent in the buffer zone and there were also signs of cattle grazing in the national sanctuary.

Data collection

We used fixed-point sampling to determine presence/absence of night monkeys in two forest habitats (Nijman and Menken 2005). Based on previous studies of the species' home range, 1.23 ha (Shanee *et al.* 2013), forty points were established approximately every 200 m along a previously established trail system. Assuming a circular home range, 1.23 ha gives a radius of 62 m, thus the 200 m interval should have avoided any overlap. Between May and July 2010, we surveyed each point five times for 20 minutes each repetition, totaling 67 hours of data collection. Sampling was carried out by pairs of observers, care was taken to minimize forest disturbance whilst accessing points and a 5-minute 'rest' was taken between arrival at each point and the start of sampling. Field assistants were trained to use the methodology and identify the night monkeys using visual and auditory cues. To avoid statistical idiosyncrasy, the surveys were divided into four time periods (18:00–21:00, 21:00–00:00, 00:00–03:00 and 03:00–06:00). We collected habitat covariate data on moon phase every day.

The vegetation at each site was characterized using the point-quarter sampling method. For each point-quarter, we measured the distance from the center point of the point-quarter to the nearest tree and subsequently to the nearest neighboring tree (DBH ≥ 10 cm) to indicate relative density, basal area and canopy cover (Krebs 1999; Sutherland 2006). Canopy cover was estimated using digital photographs of the forest canopy taken vertically from the ground. They were analyzed using Adobe Photoshop CS4 for percentage of dark pixel cover after grey scale conversion. Families and genera of trees were identified *in situ* by local guides at Cabeza del Toro and by park-guards at the Santuario Nacional Cordillera de Colán. The Shannon-Weiner and Simpson's Diversity Indexes were used as a measure of tree diversity at the genus level.

Occupancy modelling

We combined detection histories for all data points into maximum likelihood models in the program PRESENCE to obtain estimates of occupancy (ψ) and detectability (p) (MacKenzie *et al.* 2002; Bailey and Adams 2005). We modeled covariates altitude, vegetation diversity at the genus level, luminosity (percentage of illumination at the time the moon passes the meridian of the observer <www.timeanddate.com>), and canopy cover, with the data ranked using the Akaike Information Criteria (AIC). The AIC method is based on maximum likelihood estimation and encourages parsimony (models with as few parameters as necessary). The models are compared and the best has the lower AIC

value and the least number of parameters (MacKenzie *et al.* 2006). The Δ AIC is the difference between the AIC of the best model and the subsequent model. Models with Δ AIC values of <2.0 have greater support (Bailey and Adams 2005; MacKenzie *et al.* 2006).

Data collected at both sites were analyzed individually by a predefined mode, in which detection probabilities were assumed to be either constant or survey specific (MacKenzie *et al.* 2006). This resulted in a naïve occupancy estimate defined as the proportion of sites that had at least one observation of the target species for both sites and an assessment of variance in detection across the survey period. The probability of occupancy varies from 0 to 1, where 0 is complete absence and 1 is presence of target species. We analyzed estimates of the probability of occupancy and detectability as functions of different habitat covariates through a series of models by combining all data to build a likelihood function (Donovan and Hines 2007; Guillera-Aroita *et al.* 2010) (Table 1).

We evaluated all potential models and ranked them in accordance with the AIC to identify their relative contribution to occupancy and detectability (Burnham and Anderson 2002; MacKenzie *et al.* 2006). To determine which covariate or combination of covariates had the strongest support for occupancy, the sum of Akaike weights for all models in the set containing the covariate in question were summed (Guillera-Aroita *et al.* 2010). After evaluating all potential models without interaction, we selected the top eight models as having less than four AIC units from the best model (Guillera-Aroita *et al.* 2010). We used goodness-of-fit tests for selected models based on bootstrapping (1000) and Pearson's chi-squared statistic to test whether the observed frequencies of detection histories (presence/absence) had a reasonable chance of occurring. If the 95% confidence interval did not cross zero then the P value had to be less than 0.05 (threshold for significance).

Results

We detected *Aotus miconax* at 13 survey points out of 40, producing a naïve probability of occupancy of 0.33. The simplest model where occupancy and detectability probabilities were fixed, yielded estimated values of $\psi = 0.433$ and $p = 0.242$ respectively (Table 2). Observed occupancy was lower than model expectancy indicating that *A. miconax* would be present at a higher number of survey points than observed, which may be attributed to observer error, duration of survey or habitat structure.

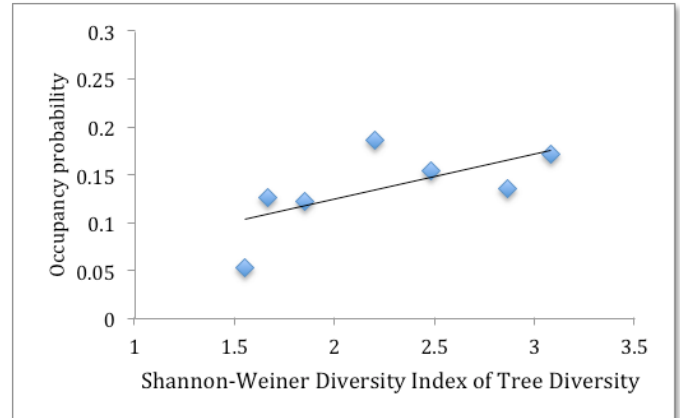
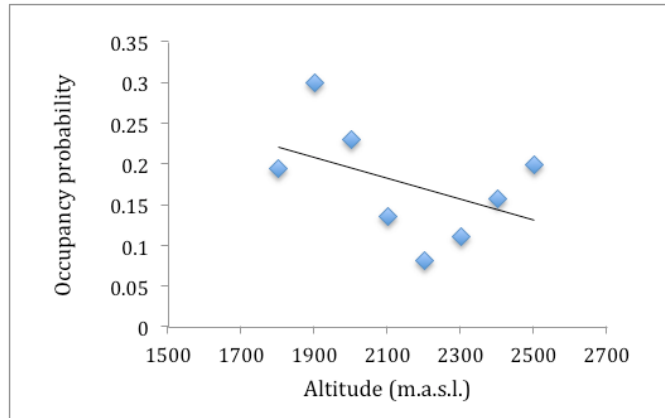
Occupancy and detectability models obtained from models with AIC less than four ranged from 128.49 for the top model to 131.92 (Table 2). The top ranked model ($\psi = 0.5054 \pm 0.87$, $p = 0.3022 \pm 0.346$ SE) showed canopy cover to be the most important covariate for occupancy and genus diversity of trees as the most important covariate for detection (Table 2). The second rated model ($\psi = 0.5066 \pm 0.42$, $p =$

Table 1 Comparison of habitat characteristics between study sites.

	Cabeza del Toro	Cordillera de Colán
Tree density (ind/ha)	790.3	396.3
DBH (cm)	190.3	247.8
Canopy (% cover)	91	84
Basal area (cm ²)	57.6	60.3
Shannon Diversity Index	1.80	2.82
Simpson's Index	0.84	0.13
Altitude (m.a.s.l.)	2.097	2.203

Table 2. Occupancy and detectability estimates obtained for models with an Akaike difference smaller than 4. The probability of occupancy (ψ) and the probability of detection (p) were modelled with covariates: canopy cover (can), genus diversity (div), altitude (alt) and luminosity of the moon (lum). **AIC:** Akaike Information Criterion, **Δ AIC:** Akaike difference, **N:** number of parameters in the model, **W_i :** Akaike weight.

	Model	AIC	Δ AIC	N	W_i	ψ	($\pm$ SE)	p	($\pm$ SE)
1	$\psi(\text{can})p(\text{div})$	128.49	0	4	0.157	0.5054		0.3062	(± 0.34)
2	$\psi(\text{alt})p(\text{div})$	128.55	0.06	4	0.152	0.5066	(± 0.42)	0.1895	(± 0.33)
3	$\psi(\text{div})p(\text{lum})$	130.00	1.51	4	0.074	0.3568	(± 0.43)	0.1197	(± 0.69)
4	$\psi(\text{can})p(\text{lum})$	130.83	2.34	4	0.049	0.5188	(± 0.01)	0.5035	(± 0.01)
5	$\psi(\text{div})p(\text{alt})$	131.31	2.82	4	0.044	0.3349	(± 0.45)	0.5902	(± 0.30)
6	$\psi(\text{alt}+\text{div})p(\text{lum})$	131.70	3.21	5	0.201	0.3531	(± 0.68)	0.5033	(± 0.01)
7	$\psi(\text{alt})p(\text{lum})$	131.87	3.38	4	0.033	0.4497	(± 0.39)	0.5034	(± 0.01)
8	$\psi(\text{can})p(\text{alt})$	131.92	3.43	4	0.039	0.5256	(± 0.01)	0.6028	(± 0.29)

**Figure 2.** Changes in the average estimate of occupancy of *Aotus miconax* as predicted by altitude and tree diversity.

0.1895 $\pm$ 0.33) also gave genus diversity as the top covariate for detection with altitude as the top covariate for occupancy (Table 2).

Comparing models incorporating covariates showed some support for the fixed model FX (ψ and p constant) with an AIC value of 130.93 (Table 2). Goodness-of-fit test based on bootstrapping and Pearson's chi-squared statistic indicated that the models fit the observed data for the FX ($\chi^2 = 20.99$, $p = 0.76$ c-hat = 0.77), the most parameterized model ($N = 6$) ($\chi^2 = 22.34$, $p = 0.68$ c-hat = 0.82) and the top model ($\chi^2 = 20.20$, $p = 0.83$ c-hat = 0.70).

Altitude and tree diversity contributed the strongest support for occupancy, and moon phase (% luminosity) was the strongest predictor for detectability (Table 2). There was a positive correlation between the probability of occupancy and the genus-level diversity of the vegetation, and a negative correlation between occupancy and altitude (Fig. 2). Model of detection probability showed that all monkeys were detected above ~30% luminosity.

Discussion

Aotus miconax is among the least-studied of the Neotropical primates (Shanee *et al.* 2013) and, with a declining population trend (Cornejo *et al.* 2019; Shanee *et al.* 2015), testing a rapid census technique that contributes to long-term monitoring of the species was the primary objective of the project. Initializing this kind of census method has many benefits to conservation programs, including straightforward and fast training for local people and volunteers, the few resources required and minimal financial input, besides being easy to adapt and replicate in different habitats and for different species.

Aotus miconax was present at 13 point surveys, giving a naïve occupancy of 0.33 over both sites. The simplest models (fixed occupancy and detectability and fixed occupancy but detectability allowed to vary over survey periods) showed an occupancy probability of 0.43 (naïve 0.45) and 0.52 (naïve 0.20) for both Cabeza del Toro and the Santuario Nacional Cordillera de Colán, respectively, suggesting that *A. miconax* was present in some sample areas but not detected. Modelling different habitat covariates allows the exploration of relationships between detection probability and habitat characteristics, and the development of predictive models of habitat preference (Guillera-Aroita *et al.* 2010). As the results indicate, the proportion of sites where a species is detected will always underestimate true occupancy in the study area when detection is imperfect (Bailey and Adams 2005). Repeated visits to a sample landscape within a relatively short time frame will reduce the likelihood of a false absence, and areas where a species has never been detected can be regarded as a genuine absence (MacKenzie 2005).

The top four ranked models show that tree diversity and percentage of moon luminosity were important factors in the detection of *A. miconax*. Fernandez-Duque *et al.* (2010) found that the nocturnal activity of night monkeys was associated with the lunar cycle, with peak activity during full moon. Similarly, Fernandez-Duque *et al.* (2008) showed that *Aotus vociferans* travelled less per night during periods of low luminosity, with an average of 495 m per night during a new moon and 795 m per night during a full moon. Similarly, slender lorises, *Loris tardigradus*, forage, travel and vocalize more during a full moon, as do southern lesser galagos, *Galago moholi* (see Bearder *et al.* 2002), and sportive lemurs, *Lepilemur mustelinus* (see Gursky 2003). Pariente (1980) quantified the amount of light from a full and new moon in a forest patch in Madagascar and found ten times more light available during a full moon. In all cases but one, *A. miconax* showed a higher rate of detection in Cabeza del Toro when moon luminosity was greater than 70%. The exception was a pair leaving their nesting site when it was still daylight. The results concur with previous research on night monkey activity patterns during the brightest part of the lunar cycle (Wright 1989; García and Braza 1993; Nowak 1999; Fernandez-Duque *et al.* 2010).

Modelling demonstrated that altitude and tree diversity were important factors influencing occupancy. Caldecott (1980) stated that as altitude increases the occupancy of a species would decrease due to changes in the forest structure: reduced floristic diversity, smaller basal area and DBH, and reduced primary production levels. Similarly, Kessler (2001) and Kroemer *et al.* (2005) found that plant species peaked in abundance at different altitudes, therefore producing a different species' composition from mid- to high altitudes. The negative correlation between the probability of occupancy and altitude, observed at both sites, therefore, indicates that the decrease in occupancy could be due to a change in vegetation composition related to biotic and abiotic factors (Smith and Killeen 1998; Bendix *et al.* 2008).

Top-ranked models in our study also highlighted genus diversity as an important factor for detectability. Cristobal-Azkarate *et al.* (2005), studying the mantled howler (*Alouatta palliata mexicana*), and Arroyo-Rodríguez *et al.* (2007), studying the Udzungwa red colobus (*Ptilocolobus gordonorum*), showed that there was a strong positive correlation between primate abundance and floristic diversity. It was expected, therefore, that the frequency of observations of *A. miconax* in the Santuario Nacional Cordillera de Colán would be higher due to a greater floristic diversity there when compared with Cabeza del Toro; however, this was not the case. All areas sampled in the Santuario Nacional Cordillera de Colán showed a higher tree genus diversity and a low Simpson's Index, meaning there were no dominant tree species, and indicating other ecological factors or a combination of factors were affecting the detectability of *A. miconax*. Incorporating different covariates in the detectability models may reveal more information that could explain our perceived discrepancies in the data.

Conclusions

Occupancy modelling with fixed-point sampling proved to be an effective method of sampling for the type of terrain and for the species. It is easy to implement in the field, requiring few resources and few people. It is also a rapid survey method that allows large areas to be assessed in a short time. Used in conjunction with multi-site density estimates and ecological niche modelling, differential occupancy models can improve the accuracy of conservation evaluations, providing realistic estimates of change in population sizes taking into account species specific responses to habitat heterogeneity and alteration. These are important considerations for future monitoring programs, as conservation effort is often constrained by financial support. One limitation of the modelling, as shown by our results, is that it is highly dependent on the covariates used. Those are species specific and can, therefore, influence occupancy and should be considered before applying the method.

The results of this study have provided useful information that will contribute to monitoring programs for *A. miconax* and other nocturnal primates. Altitude and tree diversity were key factors that influenced the probability of occupancy and tree diversity and moon phase influenced the probability of detection of *A. miconax*. Habitat and site-based management for *A. miconax* should identify tree diversity as a key habitat requirement and aim to prevent future degradation by helping to promote alternative uses of existing forests.

This study has revealed that fixed-point sampling and occupancy modelling is an effective alternative to more traditional methods of primate abundance surveys. It has been shown to be highly adaptable to a wide range of species, and we think that it has important implications for nocturnal primate research in the future, particularly for poorly known species that are difficult to detect visually.

Acknowledgments

We thank Esme Lee Hammer, Leyda Gueiler Rimachín Cayatopa, SERNAMP, Josie Chambers, Cesar Bartra, Noga Shanee and Ronal Mego Mego. The researchers complied with national laws of Peru under permit number 384-2010-AGDGFFS-DGEFFS, and the authors declare there is no conflict of interest. The work was funded by Neotropical Primate Conservation, Global Wildlife Conservation's Primate Action Fund, Primate Conservation Inc, Apenhuel Primate Conservation Trust, Le conservatoire pour la protection des primates, and Wild Futures. We also thank all the local field guides and authorities for their help.

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Received for publication: 15 January 2017

Revised: 3 July 2019