A peer-reviewed version of this preprint was published in PeerJ on 1 July 2014.

<u>View the peer-reviewed version</u> (peerj.com/articles/464), which is the preferred citable publication unless you specifically need to cite this preprint.

Cove MV, Spínola RM, Jackson VL, Saénz JC. 2014. The role of fragmentation and landscape changes in the ecological release of common nest predators in the Neotropics. PeerJ 2:e464 <u>https://doi.org/10.7717/peerj.464</u>

The role of fragmentation and landscape changes in the ecological release of common nest predators in the Neotropics

Loss of large mammalian carnivores may allow smaller mesopredators to become abundant and threaten other community members. There is considerable debate about mesopredator release and the role that other potential factors such as landscape variables and human alterations to land cover lead to increased mesopredator abundance. We used camera traps to detect four mesopredators (tayra, Eira barbara; white-nosed coati, Nasua narica; northern raccoon, Procyon *lotor*; and common opossum, *Didelphis opossum*) in a biological corridor in Costa Rica to estimate habitat covariates that influenced the species' detection and occurrence. We selected these mesopredators because as semi-arboreal species they might be common nest predators, posing a serious threat to resident and migratory songbirds. Pineapple production had a pronounced positive effect on the detectability of tayras, while forest cover had a negative effect on the detection of coatis. This suggests that abundance might be elevated due to the availability of agricultural food resources and foraging activities are concentrated in forest fragments and pineapple edge habitats. Raccoon and opossum models exhibited little influence on detection from habitat covariates. Occurrence models did not suggest any significant factors influencing site use by nest predators, revealing that all four species are habitat generalists adapted to coexisting in human altered landscapes. Furthermore, fragmentation and land cover changes may predispose nesting birds, herpetofauna, and small mammals to heightened predation risk by mesopredators in the Neotropics.

1 The role of fragmentation and landscape changes in the ecological release of common nest 2 predators in the Neotropics

- 3
- 4

6

7

8

9

Michael V. Cove*, R. Manuel Spínola, Victoria L. Jackson, and Joel Saénz 5

Department of Biology and Earth Science, University of Central Missouri, Warrensburg, MO 64093, USA (MVC & VLJ)

Instituto Internacional en Conservación y Manejo de Vida Silvestre, Universidad Nacional, Apartado 1350-3000, Heredia, COSTA RICA (RMS & JC)

*author to whom correspondence should be addressed (email: mvcove@ncsu.edu)

Current address of MVC: Department of Applied Ecology, North Carolina State University, Raleigh, North Carolina, 27695, USA

Keywords: camera traps, carnivores, coati, fragmentation, mesopredator release, Neotropics,

18 nest predators, pineapple production, raccoon, tayra

19 1. INTRODUCTION

Large carnivores receive substantial attention from the scientific community due to their 20 charismatic status and their importance as keystone species in maintaining community structure 21 (Morrison et al., 2007). Due to the loss of large carnivores from many systems, medium-sized 22 carnivores and carnivorous marsupials (collectively known as mesopredators) have recently 23 24 gained more attention (Roemer et al., 2009) because of their potential threat to migratory song birds (Crooks and Soulé, 1999; Donovan et al., 1997; Elmhagen and Rushton, 2007; Garrott et 25 26 al., 1993). The most commonly cited hypothesis for the increased abundance of mesopredators is 27 the Mesopredator Release Hypothesis (MRH –Crooks and Soulé, 1999), but there is much debate that remains over the process of mesopredator release (Cove et al., 2012a; Gerht and Clark, 28 29 2003; Elmhagen and Rushton, 2007). The MRH has support within some families, most notably 30 the Canidae, where larger canids drive the population dynamics and habitat associations of 31 smaller canids (Donadio and Buskirk, 2006; Gehrt and Clark, 2003). However, interspecific 32 killing and interference competition are less common among different families (Donadio and Buskirk, 2006). 33

The top predators in Central America are jaguars (*Panthera onca*), pumas (*Puma concolor*), 34 35 and, coyotes (*Canis latrans* – Cove et al., 2012b; Wainwright, 2007). All three predators partake in interspecific killing and may affect the distribution and habitat associations of smaller 36 37 mesopredators (Donadio and Buskirk, 2006; Palomares and Caro, 1999). The large cats are often 38 hunted due to cattle depredation and are rare. Coyotes are recent invaders and, because they prefer disturbed open habitat, are seemingly rare in many intact forested areas. Mesopredators 39 40 are common in Central America which may result from the rarity of the top predators, thus, 41 supporting the MRH and a "top down" view of their release. Another plausible explanation is

that mesopredators are better adapted to coexisting with humans in disturbed habitats and, as 42 omnivores, are able to supplement their diets with agricultural resources via a "bottom up" 43 44 release (Elmhagen and Rushton, 2007; Garrott et al., 1993; Roemer et al., 2009). Several studies in the United States attempted to model relative abundance of mesopredators 45 as functions of landscape and local habitat variables and predict predation risk for forest-nesting 46 47 birds (Crooks and Soulé, 1999; Dijack and Thompson, 2000; Donovan et al., 1997). These studies determined that mesopredator abundance, activities, and hence nest predation increased 48 49 in fragmented areas and within forest edge habitats, particularly surrounding agricultural lands. 50 However, no similar studies have examined mesopredator occurrence in Central America. This study integrated data collected from camera traps and the occupancy modeling 51 framework developed by MacKenzie et al. (2005, 2006) in order to examine habitat associations 52 53 of four common mesopredators from three different families in a fragmented biological corridor 54 in Costa Rica. We selected tayra (*Eira barbara*), white-nosed coati (*Nasua narica*), northern

raccoon (*Procyon lotor*), and common opossum (*Didelphis opossum*) as the species of interest
because they are common, adapted to human presence, and are important nest predators due to
their semi-arboreal nature.

58 **2. METHODS**

59 *Study site*

The San Juan – La Selva Biological Corridor is the northernmost portion of the
Mesoamerican Biological Corridor in Costa Rica linking the Indio-Maíz Biological Reserve of
southeastern Nicaragua to the Braulio Carrillo National Park of central Costa Rica. Although
deforestation still occurs within the corridor, government incentives (Forestry Law no. 7575)
have encouraged reforestation and tree plantations (Morse et al., 2009). Most of the land within

the corridor is privately owned with some reserves and lodges established to mitigate small scale agriculture, yet large scale pineapple plantations and cattle operations continue to expand in the corridor and surrounding landscape (Fagen et al., 2013). We selected 16 survey sites to be representative of the land cover in the corridor; selection was loosely based on accessibility and forest size. All forest sites were located on eco-lodge forest reserves, tree plantations, cattle ranches and agricultural plantations. In an effort to ensure independence among forest sites, we selected sites a minimum of 2 km apart.

72 Camera Trapping

73 We surveyed fourteen forest sites over two field seasons (July-August 2009 and June-August 74 2010), while two additional sites were surveyed from October-November 2009. Information was lost from one of these sites and was excluded from further analysis. To avoid the pitfalls of using 75 76 a single camera as a defined "site" representative of an entire forest (Efford and Dawson, 2012), 77 we decided to aggregate several traps in arrays at each site. Arrays consisted of a central camera 78 station and three additional camera stations surrounding the central station spaced at >250 m apart, for a total of four cameras in the 2009 surveys. Cameras were arranged in an array of six 79 80 spaced >250 m apart in the 2010 surveys. Previous research suggested that although this resulted 81 in variable trapnights among sites, there were no strong differences in detection as result of the 82 varying numbers of cameras (Cove et al., 2013). Each camera station consisted of a remotely 83 triggered infrared camera (Scout Guard SG550, HCO Outdoor Products, Norcross, GA, USA) or 84 a remotely triggered flash camera (Stealth Cam Sniper Pro Camera 57983, Stealth Cam, LLC, 85 Grand Prairie, TX, USA) secured to a sturdy tree 0.25-0.5 m off the ground. The camera was 86 directed at an opposing tree, 3-4 m away, baited with a secured can of sardines 1-1.5 m off the 87 ground. Although other camera trap studies set cameras along human trails and roads (Tobler et

al., 2008), we avoided areas of high human use due to threat of theft focusing on animal game
trails. Trail cameras were left at each site for 24-38 days and checked weekly for rebaiting and
battery changes.

All of our research was in accordance with the guidelines established by The American
Society of Mammalogists (Gannon et al., 2007). The camera trapping protocol was approved by
the University of Central Missouri Institutional Animal Care and Use Committee (IACUC -Permit No. 10-3202).

95 Habitat Variables and Analysis

Using ArcGIS 10.0 (ESRI, 380 New York Street, Redlands, CA 92373, USA), we overlaid
camera trap locations onto a digitized land use-land cover map. We created a 1-km radius buffer
at each site using a central point among the cameras to measure landscape covariates. Habitat
covariate selection was based on known ecology of the mesopredators and factors that might
affect their detection and occurrence (Gompper, 1995; Lotze and Anderson, 1979; McManus,
1974; Presley, 2000; Wainwright, 2007).

We measured the distance to the nearest village, creating an index of human presence and/or 102 disturbance. Forest cover is the percentage of buffer covered by primary and secondary forest 103 104 and tree plantations. Because pineapple plantations are prevalent in the region, we used a 105 binomial covariate to indicate this land use within each site buffer. The proposed Maquenque 106 National Park is also within the northern extent of the corridor and we used a similar binomial 107 covariate to denote sites as within or outside the proposed park boundaries. The final covariate was the total number of patches within each site buffer, which is an index of habitat 108 109 heterogeneity and habitat fragmentation. We standardized all continuous covariates to z scores 110 for analysis, but performed no other transformations (Long et al., 2011).

130

111 We combined all mesopredator photos from both field seasons to organize and manage binary detection histories (1= detected, 0= not detected). We partitioned detection histories into 112 five day blocks for a maximum total of seven repeat surveys per species per site. We used the 113 detection histories and habitat covariates within a single-season occupancy model in program 114 PRESENCE 2.4 (Hines, 2009). Although the data were collected over two field seasons, we did 115 not resample any sites. This analysis refers to Ψ as "site use" as opposed to "occurrence," so 116 grouping of the field seasons does not violate any of the assumptions of the modeling process 117 (MacKenzie et al., 2005).

Given our data, we developed six relatively simple *a priori* models for each species (Table 1), including a global model, to estimate the influence of habitat covariates on detection probabilities in the individual mesopredator detection models. Although detection probability is often considered a nuisance parameter, there is an apparent relationship between detection probability and local abundance because as local abundance increases the probability of 124 detecting a species will increase making it a parameter of interest in our study (O'Connell and Bailey, 2011). We did not use a seasonal covariate because all surveys were conducted during 125 the rainy season. We then used the covariates that contained high model support and had strong 126 127 effects on detection as a constant covariate set in the subsequent occurrence models (Long et al., 128 2011). For the occurrence models, we used seven *a priori* models (Table 2). 129 We determined the best approximating models based on the Akaike Information Criterion corrected for small sample size (AIC_c) and Akaike weights (ω_i). To evaluate model fit, we

performed 10,000 simulated parametric bootstraps for the global model (all covariates) to 131

132 determine if there was evidence of overdispersion (Burnham and Anderson, 2002). We considered all models contained within the 90% CI ($\sum \omega_i > 0.90$) to have substantial support as the top-ranking models (Burnham and Anderson, 2002).

135 **3. RESULTS**

From 2,286 camera trapnights, we obtained 23 independent photographs of tayras (10.06
photos/1000 trapnights), 33 photos of coati (14.44/1000 trapnights), 7 photos of raccoons
(3.06/1000 trapnights), and 23 photos of opossums (10.06/1000 trapnights). At least one nest
predator species was detected at every site, but only one site had detections of all four species.
There was no evidence of overdispersion and we evaluated all models by their AIC_c and their
Akaike weights.

Detection covariates affected each species differently (Table 3). Pineapple production had high model support ($\sum \omega_i = 0.75$) and a strong positive influence on detection probability for tayras and was used as the constant detection covariate in subsequent occurrence models. Forest cover had high model support ($\sum \omega_i = 0.67$) and a negative influence on detection probabilities for coatis and was used as the constant detection covariate in coati occurrence models. Raccoon and opossum models contained minimal support for habitat covariates influencing detection and we used a constant detection probability in the eventual occurrence models.

From the occurrence models, no covariates that we examined explained significant changes in mesopredator occurrence at the study sites (Table 4). The top-ranking models for tayra and raccoon suggested a negative influence of forest cover on both species' occurrence, but were highly variable. The constant occurrence model was top-ranking for both coati and opossum.

153 **4. DISCUSSION**

No large cats were photographed during the surveys and only a single coyote was
photographed at one site (Cove et al., 2013). Local interviews and cattle depredation were

evidence that large cats occur in the corridor; however, the sparse records suggest rarity and
precluded the use of these presence/absence data as model covariates. Therefore, we were unable
to assess the impact and influence of these top predators on the four mesopredator species
through trophic interactions, but the observed detection rates are higher than other Neotropical
studies with intact top predator communities (Tobler at al., 2008).

161 Landscape changes did affect detection parameters for the tayra and coati. Because camera traps operate 24-hr per day, heightened detection corresponds with increased local abundance or 162 increased localized activity of mesopredators as influenced by landscape covariates. Pineapple production had a strong positive influence on the detection probability of the tayra. This is most likely an effect of the additional food resources from pineapple production leading to higher local tayra abundance or concentrated foraging activities in pineapple-forest edge habitats. This may also be a consequence of concentrated foraging activities within smaller forest patches that occur in close proximity to pineapple plantations (Cove et al., 2013). The fruits not only provide 169 direct food resources to tayras but other food resources may be indirectly provided from 170 pineapple pests including small rodents, insects, and nesting birds (Presley, 2000). Pineapple production also had a positive influence on coati detection, but the effect was less pronounced. 171 172 This suggests that coati abundance is also influenced by agricultural food resources provided from pineapple production. The effect was opposite for raccoons and opossums suggesting that 173 174 pineapple production has a negative but weak influence on their detection. This may be an 175 artifact of limited raccoon detections or avoidance of areas of high use by tayras and coatis. Forest cover had a negative effect on detection probability of coatis. This suggests that coatis, 176 177 which occur in large groups, concentrate their foraging activities in small forest patches, making 178 them more easily detectable. This relationship was similar for the detection of tayras and

179 raccoons in the study area. The consequence of concentrated foraging activities in small forest patches and forest edges is also responsible for exposing nesting song birds to increased 180 predation risk (Cove et al., 2012c; Dijack and Thompson, 2000; Donovan et al., 1997). None of 181 the habitat covariates examined in this analysis were significant predictors of mesopredator 182 occurrence. The coefficients for habitat effects generally agreed with *a priori* expectations that 183 184 increased forest cover would have a negative but variable influence on occurrence. This suggests that the broad range of habitats used by these mesopredators could have drastic consequences for 185 nesting song birds, small mammals, and herpetofauna if fragmentation and forest loss continues. 186 187 Although low detections of large predators made it difficult to provide direct support for the MRH, the rarity of these species most likely plays a role in the distribution and habitat use by 188 189 mesopredators. More importantly, the compounding factors of increasing human presence, 190 decreasing forest cover, and increasing pineapple production play an important role in mesopredator release and potentially heightened local abundance. Further sampling of 191 192 mesopredator communities, as well as large predator-specific surveys and avian point count 193 surveys, will elucidate trophic interactions and the risk of predation to migratory and resident song birds. 194

195

Acknowledgements: We thank all the field assistants and lodges that helped with logistics for
this research. Special thanks to Daniel Corrales and Panthera – Costa Rica, Finca Pangola, Selva
Verde Lodge, and Laguna Lagarto Lodge for their assistance and continued support. Funding for
this research was provided by the Universidad Nacional de Costa Rica and the University of
Central Missouri International Center.

202 **REFERENCES**

- Burnham, K. P., and Anderson D. R. 2002. Model selection and multimodel inference: a
 practical information-theoretic approach. Second edition. Springer-Verlag, New York, New
 York.
- 207 Cove, M. V., Jones, B. M., Bossert, A. J., Clever Jr., D. R., Dunwoody, R. K., White, B. C. and
 208 Jackson, V. L. 2012a. Use of camera traps to examine the mesopredator release hypothesis in
 209 a fragmented Midwestern landscape. American Midland Naturalist 168: 456-465.
 - Cove, M. V., Pardo, L., Spínola, R. M., Jackson, V. L., and Saenz, J. C. 2012b. Coyote Canis latrans (Carnivora: Canidae) range extension in northeastern Costa Rica: possible explanations and consequences. Latin American Journal of Conservation 3: 82-86.
 - Cove, M. V., L. M. Niva, and V. L. Jackson. 2012c. Use of probability of detection when conducting analyses of surveys of mesopredators: A Case Study from the Ozark Highlands of Missouri. The Southwestern Naturalist 57: 258-262.
 - Cove, M. V., R. M. Spínola, V. L. Jackson, J. C. Saenz, and O. Chassot. 2013. Integrating occupancy modeling and camera-trap data to estimate medium and large mammal detection and richness in a Central American biological corridor. Tropical Conservation Science 6: 781-795.
 - Crooks, K. R., and Soulé, M. E. 1999. Mesopredator release and avifaunal extinctions in a fragmented system. Nature 400:563-566.
 - Dijack, W. D., and Thompson III, F. R. 2000. Landscape and edge effects on the distribution of mammalian predators in Missouri. Journal of Wildlife Management 64: 209-216.
- Donadio, E., and Buskirk, S. W. 2006. Diet, morphology, and interspecific killing in Carnivora.
 American Naturalist 167:524-536.
- Donovan, T. M., Jones, P. W., Annand, E. M. and Thompson III, F. R. 1997. Variation in localscale edge effects: Mechanisms and landscape context. Ecology 78:2064-2075.
- Efford, M. G., and Dawson, D. K. 2012. Occupancy in continuous habitat. Ecosphere 3:32.
- Elmhagen, B., and Rushton, S. P. 2007. Trophic control of mesopredators in the terrestrial
 ecosystems: top-down or bottom-up? Ecology Letters 10:197-206.
- Fagen, M. E., DeFries, R. S., Sensie, S. E., Arroyo, J. P., Walker, W., Soto, C., Chazdon, R. L.
 and Sanchum, A. 2013. Land cover dynamics following a deforestation ban in northern Costa
 Rica. Environmental Research Letters 8:034017.

243 244

240

229

232

- () 257 **PeerJ** PrePri
- Gannon, W. L., Sikes, R. S. and the Animal Care and Use Committee of the American Society of
 Mammalogists. 2007. Guidelines of the American Society of Mammalogists for the use of
 wild animals in research. Journal of Mammalogy 88:809-823.
 - Garrott, R. A., White, P. J. and Vanderbilt White, C. A. 1993. Overabundance: an issue for conservation biologists? Conservation Biology 7:946-949.
 - Gehrt, S. D., and Clark, W. R. 2003. Raccoons, coyotes, and reflections on the mesopredator release hypothesis. Wildlife Society Bulletin 31:836-842.
 - Gompper, M. E. 1995. Nasua narica. Mammalian Species 487:1-10.
 - Hines, J. E. 2009. Presence 2.4 software to estimate patch occupancy and related parameters: USGS-PWRC. http://www.mbr-pwrc.usgs.gov/software/presence.html
 - Long, R. A., Donovan, T. M., MacKay, P., Zielinski, W. J., and Buzas, J. S. 2011. Predicting carnivore occurrence with noninvasive surveys and occupancy modeling. Landscape Ecology 26:327-340.
 - Lotze, J.-H. and Anderson, S. 1979. Procyon lotor. Mammalian Species 119:1-8.
 - MacKenzie, D. I., Nichols, J. D., Sutton, N., Kawanishi, K., and Bailey, L. L. 2005. Improving inferences in population studies of rare species that are detected imperfectly. Ecology 86:1101-1113.
 - MacKenzie, D. I., Nichols, J. D., Royle, J. A., Pollock, K. H., Bailey, L. L. and Hines, J. E. 2006. Occupancy estimation and modeling. Academic Press, Burlington, Massachusetts.
 - 273 McManus, J. J. 1974. Didelphis Virginiana. Mammalian Species 40:1-6.
 - Morrison, J. C., Sechrest, W., Dinerstein, E., Wilcove, D. S., and Lamoreux, J. F. 2007.
 Persistence of large mammal faunas as indicators of global human impacts. Journal of Mammalogy 88:1363-1380.
 - Morse, W. C., Schedlbauer, J. L., Sensie, S. E., Finegan, B., Harvey, C. A., Hollenhorst, S. J.,
 Kavanagh, K. L., Stoian, D. and Wulfhurst, J. D. 2009. Consequences of environmental
 service payments for forest retention and recruitment in a Costa Rican biological corridor.
 Ecology and Society 14:23.
 - O'Connell, A. F., and Bailey, L. L. 2011. Inference for occupancy and occupancy dynamics.
 Pages 191-207 in O'Connell, A. F., J. D. Nichols, and K. U. Karanth, editors. Camera traps in animal ecology: methods and analyses. Springer, New York, New York.
 - O'Connell, A. F., Nichols, J. D., and Karanth, K. U. 2011. Camera traps in animal ecology:
 methods and analyses. Springer, New York, New York.

- Palomares, F., and Caro, T. M. 1999. Interspecific killing among mammalian carnivores.
 American Naturalist 153:492-508.
- Presley, S. J. 2000. Eira barbara. Mammalian Species 636: 1-6.
 - Roemer, G. W., Gompper, M. E. and Van Valkenburgh, B. 2009. The ecological role of the
 mammalian mesocarnivore. Bioscience 59:165-173.
 - Tobler, M. W., Carrillo-Percastegui, S. E., Leite Pitman, R., Mares, R., and Powell, G. 2008. An evaluation of camera traps for inventorying large and medium sized terrestrial rainforest mammals. Animal Conservation 11:169-178.
 - Wainwright, M. 2007. The mammals of Costa Rica: A natural history and field guide. Cornell University Press, Ithaca.

PeerJ PrePrints

Table 1. Descriptions and expected direction of *a priori* detection (*p*) models for mesopredators from camera trap surveys in theSan Juan - La Selva Biological Corridor, Costa Rica, 2009-2010.

Hypothesis	Model	Model Structure	Expected Result
No habitat covariates affect detection	<i>p</i> (.)	βο	-
Mesopredator abundance and foraging increase in close proximity to villages so as distance to village increases detection decreases	<i>p</i> (dist)	$\beta_0 + \beta_1(dist)$	$\beta_1 \leq 0$
Habitat heterogeneity and fragmentation increase forest edge and lead to higher foraging and detection	<i>p</i> (tnp)	$\beta_0 + \beta_1(tnp)$	$\beta_1 > 0$
Pineapple production provides food resources and increases abundance and detection	<i>p</i> (pina)	$\beta_0 + \beta_1(\text{pina})$	$\beta_1 > 0$
Increasing forest cover will have a negative effect on detection because activities will be less concentrated	<i>p</i> (for)	$\beta_0 + \beta_1(\text{for})$	$\beta_1 < 0$
Distance to village, habitat heterogeneity, pineapple production, and forest cover all affect detection	<i>p</i> (global)	$\beta_0 + \beta_1(\text{dist}) + \beta_2(\text{tnp}) \\ + \beta_3(\text{pina}) + \beta_4(\text{for})$	$\beta_1 < 0, \beta_2 > 0, \beta_3 > 0, \beta_4 < 0$

Table 2. Descriptions and expected direction of <i>a priori</i> occurrence (Ψ) models for mesopredators from camera trap
surveys in the San Juan - La Selva Biological Corridor, Costa Rica, 2009-2010.

Hypothesis	Model	Model Structure	Expected Result
No habitat effects on occurrence	Ψ(.)	β_0	_
Negative effect on occurrence within Maquenque National Park	Ψ(Maq)	$\beta_0 + \beta_1(Maq)$	$\beta_1 < 0$
Negative effect on occurrence as distance to village increases	Ψ(dist)	$\beta_0 + \beta_1(dist)$	$\beta_1 \leq 0$
Positive effect on occurrence as habitat heterogeneity increases	Ψ(tnp)	$\beta_0 + \beta_1(tnp)$	$\beta_1 > 0$
Positive effect on occurrence as forest cover increases	Ψ(for)	$\beta_0 + \beta_1(for)$	$\beta_1 > 0$
Positive effect on occurrence with pineapple production present	Ψ(pina)	$\beta_0 + \beta_1(pina)$	$\beta_1 > 0$
Maquenque National Park, distance to village, habitat heterogeneity, forest cover, and pineapple production all affect occurrence	Ψ(global)	$\beta_0 + \beta_1(Maq) + \beta_2(dist) + \beta_3(tnp) + \beta_4(for) + \beta_5(pina)$	$ \begin{array}{c} \beta_1 < 0, \ \beta_2 < 0, \\ \beta_3 > 0, \ \beta_4 > 0, \\ \beta_5 > 0 \end{array} $

1	5			Untransformed coefficients of covariates (SE)				
Species						Total number		
Model	Δ_i	ω_i	Κ	Intercept	Pineapple	of patches	Distance	Forest
Tayra								
<i>p</i> (pina)	0.00	0.611	3	-2.695 (0.662)	2.211 (0.743)	-	-	-
<i>p</i> (global)	2.97	0.138	6	-3.646 (0.837)	2.626 (0.918)	1.016 (0.423)	0.659 (0.338)	-0.303 (0.502)
<i>p</i> (tnp)	3.38	0.113	3	-1.311 (0.465)	_	0.701 (0.368)	_	_
<i>p</i> (.)	3.89	0.087	2	-0.807 (0.355)	_	_	_	_
Coati								
p(for)	0.00	0.667	3	-0.214 (0.291)	_	_	_	-0.784 (0.330)
<i>p</i> (.)	2.93	0.154	2	-0.329 (0.279)	_	_	-	_
<i>p</i> (pina)	4.26	0.079	3	-0.705 (0.409)	0.763 (0.567)	_	-	-
<i>p</i> (dist)	4.65	0.065	3	-0.313 (0.279)	-	_	-0.322 (0.271)	-
Raccoon								
<i>p</i> (.)	0.00	0.383	2	-1.157 (0.599)	-	_	-	-
<i>p</i> (tnp)	U 1.15	0.216	3	-1.793 (0.805)	_	0.691 (0.471)	_	_
<i>p</i> (dist)	2.15	0.131	3	-1.035 (0.570)	—	—	0.819 (0.747)	-
<i>p</i> (pina)	2.46	0.112	3	-0.444 (0.915)	-1.007 (1.144)	_	-	-
p(for)	2.55	0.107	3	-2.494 (1.176)	-	_	-	-0.780 (0.637)
Opossum								
p(.)	0.00	0.545	2	-0.621 (0.323)	—	—	-	-
<i>p</i> (pina)	3.08	0.117	3	-0.543 (0.402)	-0.207 (0.660)	_	-	-
p(for)	3.12	0.115	3	-0.612 (0.324)	—	—	-	0.080 (0.339)
<i>p</i> (tnp)	3.17	0.112	3	-0.628 (0.335)	-	-0.025 (0.291)	-	-
<i>p</i> (dist)	3.17	0.112	3	-0.621 (0.323)	_	_	0.039 (0.417)	-

Table 3. Selected top models and untransformed coefficients of habitat variable effects on detection probability (\hat{p}) for mesopredators from camera trap surveys in the San Juan - La Selva Biological Corridor, Costa Rica, 2009-2010.

Models presented make up the 95% confidence set, where Δ_i is AIC_c difference, ω_i is the Akaike weight, and K is the number of model parameters. Model covariates were used as a constant detection set for occurrence models for species that did not exhibit the p(.) as the top ranking model (tayra and coati).

Covariates: pina is the binomial term to identify large-scale pineapple production within the site buffer; tnp is the total number of patches within the buffer; dist is the linear distance (km) to the nearest village; for is the total percent of forest cover (primary, secondary, and tree plantations) within the site buffer.

			_	Untransformed coefficients of covariates (SE)					
Species Model	Δ_i	ω_i	K	Intercept	Forest	Maquenque	Total number of patches	Pineapples	Distance
Tayra									
Ψ (for)	0.00	0.432	4	1.568 (2.074)	-2.874 (2.709)	—	_	_	_
𝖞(Maq)	1.96	0.162	4	1.820 (1.333)	_	-2.875 (1.849)	_	_	_
Ψ(tnp)	2.14	0.148	4	1.485 (1.365)	_	—	1.507 (1.478)	_	_
𝖞(pina)	2.83	0.105	4	-0.464 (1.393)	_	—	_	2.313 (1.955)	_
Ψ(dist)	3.35	0.081	4	1.905 (1.651)	_	—	_	_	0.393 (1.144)
Coati									
Ψ(.)	0.00	0.487	3	0.532 (0.569)	_	_	_	_	_
Ψ (for)	2.63	0.131	4	0.679 (0.676)	-0.678 (0.673)	—	_	-	-
𝖞(Maq)	2.77	0.122	4	0.055 (0.727)	_	1.247 (1.307)	_	_	_
Ψ(tnp)	2.95	0.111	4	0.528 (0.579)	_	—	0.593 (0.667)	_	_
Ψ(dist)	3.74	0.075	4	0.544 (0.577)	_	_	_	_	0.162 (0.576)

Table 4. Selected top models and untransformed coefficients of habitat variable effects on occurrence models (Ψ) for mesopredators from camera trap surveys in the San Juan - La Selva Biological Corridor, Costa Rica, 2009-2010.

Models presented make up the 90% confidence set, where Δ_i is AIC_c difference, ω_i is the Akaike weight, and K is the number of model parameters. Coefficients are in logit space and relate to standardized covariates.

Covariates: for is the total percent of forest cover (primary, secondary, and tree plantations) within the site buffer; Maq is the binomial term for sites within the proposed Maquenque National Park; tnp is the total number of patches within the buffer; pina is the binomial term to identify large-scale pineapple production within the site buffer; dist is the linear distance (km) to the nearest village.

				Untransformed coefficients of covariates (SE)					
Species Model	Δ_i	ω_i	K	Intercept	Forest	Maquenque	Total number of patches	Pineapples	Distance
Raccoon									
Ψ (for)	0.00	0.623	3	-1.940 (1.218)	-2.052 (1.203)	—	_	-	-
Ψ(.)	2.52	0.177	2	-1.129 (0.729)	—	—	_	-	-
Ψ(pina)	4.64	0.061	3	-1.845 (1.115))	—	—	_	1.444 (1.458)	-
Ψ(tnp)	4.69	0.060	3	-1.270 (0.789)	—	—	0.699 (0.718)	-	-
Opossum									
Ψ(.)	0.00	0.456	2	0.276 (0.569)	—	—	_	-	-
Ψ(tnp)	2.11	0.159	3	0.287 (0.594)	—	_	-0.600 (0.604)	-	_
Ψ (for)	3.07	0.098	3	0.278 (0.573)	-0.198 (0.600)	—	_	_	-
𝖞(Maq)	3.08	0 <mark>.09</mark> 8	3	0.118 (0.757)	—	0.349 (1.133)	_	_	-
𝖞(pina)	3.13	0.096	3	0.381 (0.743)	—	—	_	-0.258 (1.138)	-

Table 4 continued.

Models presented make up the 90% confidence set, where Δ_i is AIC_c difference, ω_i is the Akaike weight, and K is the number of model parameters. Coefficients are in logit space and relate to standardized covariates.

Covariates: for is the total percent of forest cover (primary, secondary, and tree plantations) within the site buffer; Maq is the binomial term for sites within the proposed Maquenque National Park; tnp is the total number of patches within the buffer; pina is the binomial term to identify large-scale pineapple production within the site buffer; dist is the linear distance (km) to the nearest village.

PeerJ PrePrints