

# Raptors of A Neotropical City: Diversity And Habitat Relationships Along An Urbanization Gradient

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# Abstract

Urbanization involves changes in landscape terrain, hydrology, and vegetation. These changes allow some wildlife species to thrive in cities while blocking others. We analyzed how a gradient of urbanization in the city of Xalapa could have a filtering effect that prevents some raptors from occurring or effectively use the most urbanized areas, and where such filtering effects occur. We selected 6 habitat variables that could predict the detection and occupation of raptors along the urban gradient. We conducted direct observations and call-broadcast surveys to estimate relative abundance, richness, and diversity of raptors from June 2019–February 2020. We visited 20 sites during three seasons, from dawn–11:00 and 18:00–23:00 h. We obtained 201 individual records of 14 species. The best-preserved part of the gradient (< 7.7% urban cover) had the highest species diversity, while the 4th category along the gradient (53.9% urban cover) had the lowest, suggesting that at this level of urbanization an abrupt filtering effect occurs on the raptor community. The proportion of exotic/native plant species was the best detection predictor for most of the diurnal species and *Ciccaba virgata*. The proportion of urban relative to green area was negatively associated with the occupation of *Accipiter cooperii* and *Rupornis magnirostris*. Our results suggests that this Neotropical city effectively acts as a filter for most species, with only 4 raptors able to use or occupy most of the urban gradient. Our work represents a critical first step towards understanding how the process of urbanization influences a raptor community.

## Introduction

Our planet is urbanizing rapidly, as the Earth's urban-dwelling human population grows at a rate of one million people per week (Grimm et al. 2008). Urban areas represent one of the most drastic manifestations of anthropogenic habitat modification, seriously altering nutrient cycles, hydrology, heat balance, and different land-use changes (Ramírez-Bastida 2000). Urbanization has become one of the main threats to global biodiversity (García-Frapolli and Toledo 2008; McKinney 2008; Sulaiman et al. 2013).

Birds are one of the most commonly used biological groups to assess the impact of urbanization on biodiversity because of their sensitivity to environmental changes (Moreno et al. 2007; McKinney 2008; Shochat et al. 2010). Some birds are less tolerant to changes in land use, and anthropogenic environmental effects such as pollution, noise, traffic, and even the occurrence of domestic animals can affect their presence. For example, Northern harriers (*Circus hudsonius*), ferruginous hawks (*Buteo regalis*), and rough-legged hawks (*Buteo lagopus*) are very rare or not detected at all in cities (Díaz and Armesto 2003; Boal and Dykstra 2018), whereas others, in turn, are relatively tolerant to changes caused by environmental transformations related to urbanization (Chace and Walsh 2006; Filloy and Bellocq 2007).

In North America, Cooper's hawks (*Accipiter cooperii*) have been reported living in forest and suburban areas in cities across the United States. Peregrine falcons (*Falco peregrinus*) have been reported nesting in New York City's skyscrapers and Harris's hawks (*Parabuteo unicinctus*) have become a regular residents of Mexico City (Chace and Walsh 2006; Filloy and Bellocq 2007; Ortega-Álvarez and Calderón-Parra 2014, p.; Boal 2018). In tropical and subtropical regions, some raptors such as Mississippi kites (*Ictinia mississippiensis*) and ferruginous pygmy-owls (*Glaucidium brasilianum*) are known beneficiaries of urbanization (Seress and Liker 2015; Boal 2018).

Urban areas generally provide more stable environmental conditions than natural areas (e.g. permanent water sources, abundant potential nesting sites, fewer predators, and abundant prey; Gehlbach 1996). For this reason, the demographics of some raptors living in cities should reflect those of animals living in high-quality habitats. Mississippi kites, and merlins (*Falco columbarius*) in the United States, for example, have managed to increase their populations in urban areas (Boal 2018). Additionally, heat islands generated in urban areas can help increase the productivity of raptor populations due to the increased availability of ectothermic and endothermic prey (Gehlbach 1996).

In recent years, there has been a growing interest in studying how birds respond to land-use change along urban gradients, where cities tend to spread (Marzluff 2001; Chace and Walsh 2006; Croci et al. 2008; McKinney 2008; Hager 2009; Rullman and Marzluff 2014). Urbanization involves the modification or replacement of pre-existing systems. Their boundaries can act as filters that prevent the passage of some bird species due to their inability to use resources therein or adapt to urban conditions (Croci et al. 2008; Fuller et al. 2009; Sushinsky et al. 2013; Sol et al. 2014; MacGregor-Fors and Escobar-Ibáñez 2017).

Urbanization tends to favor granivores, aerial insectivores, ground-foraging insectivores, and residents over migrants (Chace and Walsh 2006). Urban-adapted birds seem to prefer forested, meadow, and cliffs habitats than shrub, aquatic, or open spaces within the urban matrix (Croci et al. 2008). Some species prefer to nest under exotic plant species due to their greater protective cover than native plants. In studies where the urban gradient is more finely divided, the peak of avian diversity is found in areas of moderate levels of disturbance, often in suburban areas or at the urban/wildland interface (Chace and Walsh 2006).

The aim of our work was to determine how a gradient of urbanization in the city of Xalapa-Enriquez (referred to as Xalapa hereafter), could have a filtering effect on raptor species that differentially prevents some species from approaching the most urbanized areas while allowing others, and at what point along this gradient the filtering effect has reached its most abrupt change. We are also interested in determining which habitat variables are predictive of the urbanization gradient occupation.

We hypothesize that the city of Xalapa exerts a differential filtering effect on the raptor community along an urbanized gradient. We predict that raptor diversity will be lower in the most urbanized areas of this gradient, with urbanized habitat negatively influencing the presence and occupancy of these birds. We also expect a progression in raptor diversity in areas with a higher proportion of vegetation cover, and consequently the highest diversity values on areas with the highest proportion of vegetation cover.

## Methods

### Study area

The city of Xalapa is located in central Veracruz, Mexico (19° 31'–19° 36' North, 96° 55'–96° 59' West, elevation 1120–1720 m; rainfall 1100–1600 mm/year). The urbanized area of this city is ~64 km<sup>2</sup> and has 488,531 inhabitants (INEGI 2020). Botanists have reported 140 tree and shrub species in the streetscape of Xalapa, with exotics representing more than half of the recorded species, and native species distributed unevenly throughout the city. Among the most conspicuous tree species are weeping fig (*Ficus benjamina*), Chinese hibiscus (*Hibiscus rosa-sinensis*), paper flower (*Bougainvillea glabra*), rhododendron (*Rhododendron* sp.), Mexican

cypress (*Cupressus lusitanica*), and little-leaf boxwood (*Buxus microphylla*). Despite its exponential growth and substantial vegetation loss, Xalapa is home to more than 340 bird species that includes 31 raptor species (González-García et al. 2014, 2016), and is considered as a natural laboratory for ecological studies by urban ecologists (MacGregor-Fors et al. 2015). Recent studies show that Xalapa's woodland covers around 40% of its territory (Falfán et al. 2018). Land uses in Xalapa include cloud forest remnants, coffee, sugarcane and corn fields, grasslands, wetlands, water bodies, and urban areas (Benítez Badillo 2011; Lemoine-Rodríguez et al. 2019).

## Urbanization gradient and sampling sites

To characterize the urbanization gradient, the city's vegetation was categorized based on the Normalized Difference Vegetation Index (NDVI) from a 2019 Sentinel 2A satellite image (ESA 2019). The NDVI allows for the generation of an image showing the greenness (vegetation reflectance indicative of relative plant biomass). This index takes advantage of the contrasting characteristics of two bands of a multispectral image: the chlorophyll pigment absorptions reflect in the red band and the high plant reflectivity in the near-infrared band (Nageswara Rao et al. 2005). This process was performed in ArcMap 10 (ESRI 2012) using the NDVI tool from the Image Analysis module.

First, it was necessary to join the spectral bands of the Sentinel satellite image. For this purpose, we used bands 2, 3, 4, and 8 with a spatial resolution of 10 m. Second, we cropped the resulting image over the urban area of Xalapa at a scale of 1:68,000 to exclude areas that were not of interest for the analysis. We generated a polygon at a scale of 1:80,000 to make the NDVI corresponding to the urban boundary of Xalapa that was modified from the polygon of Lemoine-Rodríguez et al. (2019) and INEGI (2010).

The highest percentage of urbanization obtained from the NDVI was 77% since this city's gray, impervious surface has interspersed vegetation on city parks, ravines, ridges, and sidewalks. For this reason, the NDVI does not reach 100%. From this urbanization continuous value, we generated 10 categories of the gradient (from 1–10), where 1 is 77% urban cover, and 10 is <7.7% urban cover. It is worth mentioning that we decided to include in the 10<sup>th</sup> category two preserved areas completely away from the urbanization of the city to obtain the highest possible NDVI greener value. Subsequently, we generated 2 random sampling sites in ArcMap for each urban gradient category that were at least 2 km apart (20 sampling sites in total; Figure 1). For each sampling site, we delimited a 500 m radius around the center using ArcMap 10, in order to characterize the surface composition (in ha) of the urbanized to green area of each site.

## Raptor surveys

A pilot test was carried out during May 2019 in the selected sampling sites in order to evaluate the logistics of the project and preliminarily detect which raptors were present. We visited each site two times during three field seasons: breeding (from June–July 2019), fall migration (from August–November 2019), and winter (from December 2019–February 2020). We surveyed each site twice a day, from dawn–11:00 and from 18:00–23:00 h (Bird et al. 2018).

For diurnal and nocturnal raptor censuses, we conducted systematic sampling at each of the selected 500 m radius circular plots (Bildstein and Bird 2007). For species identification, we used (Clark and Schmitt 2017), cellphone applications such as iNaturalista (CONABIO 2019) and Merlin (The Cornell Lab of Ornithology 2020), 10×42mm Celestron Nature Dx binoculars, and a Nikon D7500 camera with a 200–500 mm lens. Whenever possible, we took photographs to document our records.

For nocturnal raptors, whose study is difficult because of their nocturnal and elusive habits that make direct binocular observations impossible (Aguilar et al. 2001), we used audio playback recordings. We obtained recordings from the xeno-canto.org repository (Hardy and Morrison 2000; Enríquez and Rangel-Salazar 2001) and the Merlin application for cellphones. When possible, we used flashlights to allow the observation and identification of the species.

At each observation site we recorded the site number, urban gradient category of the site, survey number, day of the survey (hereafter “day”; recorded as an ordinal number), time of the day (diurnal or nocturnal), site coordinates, altitude, raptor season (breeding, fall migration, and winter), estimated vegetation proportion (native and exotic), presence or absence of water bodies, pedestrians, pets, raptor species, type of detection (visual or auditive), detection/non-detection of the species, behavior (hunting, perched, nesting, courtship, flying, and gliding), and perch type used (tree or building; Berthiaume et al. 2008; Conway et al. 2009; Dowling et al. 2012). To document pedestrians and pets at each sampling site, we assigned a subjective category from 0–4 (e.g., pedestrians: where 0 was the absence of the variable, 2 will be moderate, and 4 intense activity; Hogg and Nilon 2015).

## Data analyses

A widely used sampling design to determine the factors or habitat characteristics that determine the presence of species in urban areas is to repeatedly survey a selection of sampling units and record detections and no detections (Rota et al. 2016). Failure to detect a species may be due to the species being truly absent from the sampling unit or simply remaining undetected (MacKenzie et al. 2002). Binomial zero-inflation models (hereafter referred to as occupancy models; MacKenzie et al. 2017) are useful in modeling the factors that influence the probabilities of species occupancy in the face of imperfect detection (Melles et al. 2003; Kroll et al. 2007; Henneman and Andersen 2009; Hansen et al. 2011; Hogg and Nilon 2015; Rota et al. 2016).

We assigned each species to an abundance category that expresses the probability of detection of the species, according to Pugnali and Chamorro (2006). We also calculated relative abundance (RelAb) with the number of observed and heard individuals of each species divided by the number of observed individuals of all species at each sampling site ( $\text{RelAb} = \frac{\#ind\_sp}{\#ind\_total \text{ per site}}$ ). The categories were accidental: 0–1%, rare: 1–9%, scarce: 10–30%, common: 31–64%, frequent: 65–89% and abundant: 90–100%). This result, multiplied by 100 ( $\text{RelAb} \times 100$ ) corresponds to the percentage that each species contributes to the community (Smith and Smith 2001).

We used the Bray Curtis distance measurement (QS) to obtain the dissimilarity values between sampling sites and urbanization gradient. When used in presence/absence data, the Bray Curtis index is known as Sorensen's measured distance. QS is the similarity ratio and varies from 0–1. This expression easily extends to abundance

rather than the presence/absence of species. The value 0 indicates full dissimilarity between the areas sampled because they do not share species, and the value 1 indicates total similarity; intermediate values are considered as <0.25 Low, 0.26–0.5 Moderate, 0.51–0.75 High, and 0.76–1 Total similarity (Bray and Curtis 1957). Raptor diversity along the urban gradient was calculated using Shannon Exponential Index ( $e^H$ ), which gives all species an equal weight to their relative abundance, without favoring or penalizing either rare or frequent species:  $q = 1$ . This index is expressed as a positive number, which in most ecosystems vary between 0.5–5, although its normal value is between 2–3; values <2 are considered low, and >3 are high (Jost and González-Oreja 2012).

## Modeling habitat associations

We used sampling data from each site to construct detection probability and occupancy probability models in a two-stage theoretical information modeling approach (Hansen et al. 2011; MacKenzie et al. 2017). In the first stage, we modeled the detection probability for each sampling site with each of the variables recorded per site. In the second stage, the occupancy probability was modeled by species at each sampling site by incorporating habitat characteristics to find the best fit for each species. We analyzed the candidate models with the Unmarked library in R (Fiske and Chandler 2011) and the best models were selected using the Akaike Information Criterion with correction for small samples (AICc). We also calculated Akaike's weight for each model ( $w_i$ ).

Following that, we calculated the over-dispersion parameter ( $\hat{c}$ ) over 1000 parametric bootstraps of the R-occupancy data. This procedure generates 1000 random detection data sets and adjusts the global model for each of them by producing 1000 statistical Chi-square ( $\chi^2$ ) tests. The P-value of this procedure refers to the probability that each statistical test is greater than or equal to the chi-square value calculated by the global model adjusted to the currently observed dataset ( $\chi^2_{obs}$ ). The dispersion parameter ( $\hat{c}$ ) is the average of all statistical Bootstrap tests divided by the observed statistical tests ( $\chi^2_{obs} / \chi^2$ ). A value of  $\hat{c}$  near 1 was considered to be a well-fitting model for the data (Hogg and Nilon 2015). A value  $\hat{c}$  greater than 1 indicated that the data were overestimated, or that there was greater variation in the observed data than the overall model fitted to the species' occupation. We modeled the probability of detection with the variables that were considered that could influence the presence of these birds at each sampling site: day, people, pets, and proportion of native/exotic vegetation per site.

The second stage involved the creation of a set of candidate models to test the effects of measured habitat variables on observed variation in occupation ( $\psi$ ) between sampling sites, given the modeled detection probabilities (Kroll et al. 2007; Henneman and Andersen 2009; Hansen et al. 2011). The habitat variables used were altitude, NDVI, presence/absence of water bodies, and composition of the 500 m radius sampling sites (urban/vegetation area in hectares).

## Results

### Species richness and abundance

We recorded 201 individuals of 14 species of raptors along an urbanization gradient in Xalapa during the three field seasons. These species are distributed in the families Accipitridae (6 species), Falconidae (4 species), Strigidae (2 species), Tytonidae (1 species), and Pandionidae (1 species).

The Sorensen index showed very low similarity between the most urbanized gradient categories (1 and 2:  $QS=0.176$ ; 1 and 3:  $QS=0.257$ ) sharing three and two species each respectively, as well as sites 3 and 6 ( $QS=0.191$ ) sharing two species. The rest of the urban gradient had moderate to high similarity (Table 1). The best-preserved sites within the urban gradient (NDVI values of 9 and 10) had the highest Shannon exponential indexes, with seven species ( ${}^1D = 6.335$ ) and ten species ( ${}^1D = 7.022$ ) respectively, while the sites in NDVI 3 ( ${}^1D = 1.984$ ) and NDVI 4 ( ${}^1D = 1.00$ ) had the lowest Shannon indexes observed (Table 2), this is where we consider the filter of species to be occurring.

The most abundant species throughout our study were the mottled owl (*Ciccaba virgata*, RelAb=38.81%) and the roadside hawk (*Rupornis magnirostris*, RelAb=17.91%). The least abundant was the peregrine falcon (RelAb=0.50%) with only one record. Five species are legally protected (Pr) by Mexican law's NOM-059 (citation to NOM-059; Table 3).

From our total number of records, 128 individuals of nine species were vocalizing, 28 individuals from 11 species were observed flying, 29 individuals from 10 species were perched in trees and only three mottled owls were observed perching in buildings. We observed two owl species hunting (two ferruginous pygmy-owls and one *mottled owl*) in the two most urbanized categories of the gradient, as well as one osprey (*Pandion haliaetus*), five ferruginous pygmy-owls, and one *roadside hawk* individuals hunting within the middle urban gradient (categories 5 and 6). We only observed hook-billed kites (*Chondrohierax uncinatus*) displaying territorial behavior in every site they were found (n=4 sites), from half the urbanization gradient to the best-preserved places.

Mottled owls and ferruginous pygmy-owls were the most widely distributed and abundant species along the urban gradient in Xalapa. The mottled owls were observed in 9 of the 10 gradient categories whereas the ferruginous pygmy-owls were distributed in 8 of the 10 categories. Regarding diurnal species, roadside hawks and Cooper's hawks were the most widely distributed Accipitridae along the urban gradient. Although roadside hawk was the most abundant species, its presence was limited from half the urbanization gradient and upwards (NDVI>5), however, the Cooper's hawk was the only species from this family observed in the most urbanized category of the gradient (NDVI 1). The rest of the accipitrid species were filtered out from half of the gradient upwards (NDVI>5). Falconids were observed far from the urban sites, in the most conserved plots of our urban gradient (NDVI 9 and 10) with very low abundances, except for bat falcons (*Falco ruficularis*), who were observed in urban and conserved sites alike. It is important to mention that in the more urbanized sites (NDVI values 1–5), species richness and abundance tends to be more homogeneous, with only 2 or 3 species being more abundant than others, while the best-preserved sites (NDVI 7, 9, and 10), have greater species richness and abundance of individuals such as Cooper's hawks, roadside hawks, ospreys, bat falcons and collared forest-falcons (*Micrastur semitorquatus*, Figure 2).

## Detection models

Detection and occupation models were carried out for 10 of the 14 species (Table 4). We used multi-season occupancy models for each species considering the breeding, fall migration and winter seasons between June 2018–February 2019. It was not possible to make detection and occupation models for the short-tailed hawk (*Buteo brachyurus*), peregrine falcon, American kestrel (*Falco sparverius*) and barn owl (*Tyto alba*) due to the few observations made.

The best adjusted model for the detection of Cooper's hawks, broad-winged hawks (*Buteo platypterus*), roadside hawks, bat falcons, and collared forest-falcons incorporated the proportion of exotic/native plants found per site (Table 4), which indicated that the models were positively associated with the detection of all species when more native plant species were present (Figure 3). Regarding hook-billed kites, the best detection model incorporated the presence/absence of pets, and for mottled owls the combination of the day of sampling and the exotic/native plant proportion, although the latter two models should be taken with caution due to the high overestimation in the model predictions (Table 4). Pets' presence was negatively associated to the detection of hook-billed kites. For mottled owls the proportion of native plant species was negatively associated with its detection and the probability of detection was higher in the first half of the year (Figure 3). For gray hawks (*Buteo plagiatus*), ospreys, and ferruginous pygmy-owls, the null model (without field variables) was the best predictor for species detection.

## Occupancy models

Upon obtaining the best detection models, we generated occupancy models for each species. We also included the species with null detection models (without detection variables; Table 5). For Cooper's hawks and roadside hawks, the urban surface cover with the detection variables (proportion of exotic/native plant species) were considered the best predictors for species occupation (Table 5). Urban land cover was negatively associated with the predicted occupancy for both species, and over 40 ha of surface reduced the predicted occupancy for both species (Figure 4). For broad-winged hawks, hook-billed kites, bat falcons, and collared forest-falcons, the null model (without occupancy variables) and the best detection model for each species (proportion of exotic/native plant species) were the best predictor of species occupancy. For mottled owls, the best detection model was the variables included in the detection model (*i.e.*, with day+proportion of exotic/native plant species) (Figure 4). We were not able to generate occupancy models for gray hawks, ospreys, and ferruginous pygmy-owls with the variables considered for this project.

## Discussion

We observed 14 of the 31 raptors that have been recorded in Xalapa (González-García et al. 2014, 2016), which is equivalent to 45% of all the species known to occur in this city. The most abundant family was Accipitridae, which is consistent with the work carried out in the same city by the aforementioned authors; however, we found 10 fewer species within this family. This difference in the total raptor species observed may be because we randomly selected sites to cover the entire urbanization gradient rather than the complete sites where previous work has taken place.

Our results showed that there is an abrupt filtering effect occurring within the urban gradient, with NDVI categories 1–4 having less than three species, dominated mainly by mottled owls. This represents a decrease

of more than 70% of the species that were found through the urban gradient, which is consistent with previous studies where the number of species at the urban core is reduced to less than half of that found in the rural or more natural areas (Blair 2001). We consider the 4<sup>th</sup> category of the gradient as the threshold where the most noticeable filtering effect happened, because we only observed mottled owls during all three seasons.

From this 4<sup>th</sup> NDVI category to the most urban gradient (NDVI 1), we observed only four of the 14 species (Cooper's hawks, mottled owls, *bat falcons*, and ferruginous pygmy-owls). Behavioral, physiological, and ecological plasticity may contribute to an urban bird's ability to tolerate a broad array of environmental conditions, including highly disturbed, urban habitat. This plasticity traits may include a bird's ability to adjust behavior in response to novel conditions, to resist detrimental physiological effects of breeding in an urban habitat, or its proneness to use novel resources, such as food types or nesting sites (Bonier et al. 2007). Thus, we considered them urban exploiters (McKinney 2002, 2008).

Information on raptors in cities has been little studied outside North America and Western Europe. In Xalapa, a Neotropical city, there are no studies on the abundance of raptors in this city. What we know about them is limited to presence records. Our results showed that the roadside hawk was the most abundant and widely distributed diurnal raptor, which is similar to the findings of protected, wild habitats of the Los Tuxtlas Biosphere Reserve in southern Veracruz (Labra and Escalante 2013) and in rural areas of Brazil (Barros et al. 2010). Bat falcons were the most abundant falcons in our study; however, in natural areas is not very abundant (Ramírez-Bastida 2000; Vázquez-Pérez et al. 2009; Labra and Escalante 2013; Carmona et al. 2017). Regarding owls, the mottled owls and ferruginous pygmy-owls were the most abundant and most widely distributed nocturnal raptors in our study, which is consistent with studies in natural areas of Chiapas (Rivera-Rivera et al. 2012), and urban areas of Hidalgo (Valencia-Herverth et al. 2012). It is worth noticing that the most widely distributed and abundant nocturnal raptors in Mexico are barn owls, burrowing owls (*Athene cunicularia*), and great horned owls (*Bubo virginianus*; Enríquez 2017), a result that contrasts with our findings in Xalapa, where we only observed two barn owls during all three seasons.

According to Cavicchia and García (2012), the roadside hawk is a generalist and opportunistic species whose ample diet allows them to take advantage of diverse food sources, which explains to a certain degree its presence in both green and urbanized sites. Although it has been considered as an urban exploiter in rural areas of Brazil (Barros et al. 2010), in our work we can consider them as urban adapter for Xalapa (Marzluff 2001) because it was found from half of the gradient upwards (NDVI>5) and mainly looking for food in green spaces. We found that mottled owls can inhabit sites with <25% of green cover which differs from the findings of Marín-Gómez et al. (2020). They showed that the presence of this species in the same city was positively related to areas with >25% of green. This difference in results may be due to the detection method, as we made direct observations and audio provocation within a radius of 500 m, but they studied this owl through the vocal activity of the species with autonomous recording units, within 50 m radius buffers.

The urban cover showed a negative association with the predicted occupancy for Cooper's hawks and roadside hawks, suggesting that these birds will have a higher occupancy probability when there are more native species and greener surface. Regarding occupancy models for urban raptors in this city or even the country, there were no published occupancy estimates or detection probabilities in the literature to directly compare to our study. However, for Cooper's hawks is known that occupation was positively associated to woodland cover in some

US cities (Boal 2018), as well as prey abundance (Rullman and Marzluff 2014). For the rest of the species used in the detection models, the null occupation model (without occupation variables) was the best predictor for the probability of species occupation. We were not able to generate occupancy models for gray hawks, ospreys, and ferruginous pygmy-owls with the variables considered for this project, and in the case of short-tailed hawks, peregrine falcons, American kestrels, and barn owls due to the few observations made.

In conclusion, our results support our hypothesis that there is a differential filtering effect acting on diurnal and nocturnal raptors. Only 5 species are able to use or occupy the higher urbanization categories of the city, hence, providing valuable information toward understanding how the process of urbanization influences the raptor community of this city. Because this was the first study to assess how a Neotropical city like Xalapa can act as a filter to species of the raptor community, we recommend that future studies should include relevant variables that can emphasize the reasons why this is happening, such as prey availability, perching sites, temperature, ambient noise, tree cover, canopy height, arboreal cavities, among others.

## **Declarations**

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### **Conflicts of interest/Competing interests**

Each named author has substantially contributed to conducting the underlying research, drafting this manuscript and approved its latest version. Additionally, to the best of our knowledge, the named authors have no conflict of interest, financial or otherwise.

### **Availability of data and material**

The authors confirm that the data supporting the findings of this study are available within the article.

### **Code availability**

Not applicable

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## Tables

Table 1. Bray-Curtis dissimilarity index (Sorensen) for raptor diversity along the urban gradient (NDVI) in Xalapa. Decimal number represents Sorensen index between urban gradient categories. Numbers in bold are species richness per site. Non-decimal numbers are shared species between gradient categories.

NDVI	1	2	3	4	5	6	7	8	9	10
<b>1</b>	<b>3</b>	2	2	1	2	2	2	2	2	3
<b>2</b>	0.176	<b>3</b>	2	1	3	2	3	2	2	3
<b>3</b>	0.257	0.297	<b>2</b>	1	2	2	2	1	1	2
<b>4</b>	0.810	0.739	0.667	<b>1</b>	1	1	1	1	0	1
<b>5</b>	0.476	0.409	0.422	0.677	<b>7</b>	3	6	3	3	6
<b>6</b>	0.364	0.391	0.191	0.697	0.407	<b>5</b>	4	2	2	4
<b>7</b>	0.588	0.571	0.634	0.825	0.410	0.525	<b>9</b>	4	4	7
<b>8</b>	0.389	0.368	0.385	0.600	0.304	0.333	0.472	<b>4</b>	3	4
<b>9</b>	0.833	0.737	0.795	0.680	0.609	0.667	0.583	0.450	<b>7</b>	5
<b>10</b>	0.676	0.657	0.634	0.825	0.564	0.550	0.346	0.528	0.500	<b>10</b>

Table 2. Shannon exponential index for raptor diversity along the urban gradient (NDVI) in Xalapa, Veracruz, Mexico, a Neotropical city. The category of the gradient with lowest diversity is marked in bold.

NDVI	Urban Cover (%)	Richness	Abundance	Shannon_Exp (1-D)	Simpson (1-D)
1	77.7	3	15	2.074	0.417
2	69.3	3	16	2.086	0.406
3	6.6	2	16	1.984	0.492
<b>4</b>	<b>53.9</b>	<b>1</b>	<b>1</b>	<b>1.000</b>	<b>0.000</b>
5	46.2	7	21	4.594	0.716
6	38.5	5	22	3.404	0.648
7	30.8	9	45	6.215	0.800
8	23.1	4	12	2.309	0.416
9	15.4	7	11	6.335	0.826
10	<7.7	10	42	7.022	0.809

Table 3. Raptor species observed in Xalapa, Veracruz, Mexico, a Neotropical city, during the three sampling seasons from June 2018 to February 2019. For each species its seasonality is shown, if they belong to any risk category in NOM-059-SEMARNAT-2010, the count of individuals for each season and the total observed individuals, its relative abundance (RelAb) and corresponding category of abundance.

Species	Seasonality	NOM 059	Breeding	Fall Migration	Winter	Total	RelAb (%)	Abundance
<b>Accipitridae</b>								
<i>Accipiter cooperii</i>	M/W/R	Pr	1	3	5	9	4.48	Rare
<i>Buteo brachyurus</i>	O		0	1	1	2	1.00	Rare
<i>Buteo plagiatus</i>	O		2	2	1	5	2.49	Rare
<i>Buteo platypterus</i>	M	Pr	0	2	1	3	1.49	Rare
<i>Chondrohierax uncinatus</i>	M/W	Pr	0	4	6	10	4.98	Rare
<i>Rupornis magnirostris</i>	R		12	12	12	36	17.91	Scarce
<b>Falconidae</b>								
<i>Falco peregrinus</i>	W/M/R	Pr	1	0	0	1	0.50	Accidental
<i>Falco ruficularis</i>	R		1	5	4	10	4.98	Rare
<i>Falco sparverius</i>	M/W		0	2	0	2	1.00	Rare
<i>Micrastur semitorquatus</i>	R	Pr	4	0	2	6	2.99	Rare
<b>Pandionidae</b>								
<i>Pandion haliaetus</i>	M/W		0	5	5	10	4.98	Rare
<b>Strigidae</b>								
<i>Ciccaba virgata</i>	R		44	7	27	78	38.81	Common
<i>Glaucidium brasilianum</i>	R		6	7	14	27	13.43	Scarce
<b>Tytonidae</b>								
<i>Tyto alba</i>	R		0	1	1	2	1.00	Rare
Total observation per season			71	51	79	201		
Number of species registered per season			8	12	12	14		

Nomenclature is based on the checklist of the American Ornithologists' Union (AOU 1998) up to its last supplement (Chesser et al. 2020). Seasonality: R = resident, M = fall migrant, W= winter resident. Conservation categories: NOM-059 (SEMARNAT 2010; abbreviations as published), Pr = special protection.

Table 4. Best detection models for the 10 selected species, classified by the Akaike Information Criterion for small samples (AICc). DAICc is the difference between the AICc value of the highest model.

Species	Models	AICc	DAICc	$w_i$	k	LL	$\hat{c}$	P-value
<b>Accipitridae</b>								
<i>Accipiter cooperii</i>	y (.),p (Spveg.Exo/Nat)	81.97	0.00	0.50	5.00	-33.84	0.59	0.778
<i>Buteo plagiatus</i>	y (.),p (Null)	46.04	0.00	0.22	4.00	-17.69	0.86	0.51
<i>Buteo platypterus</i>	y (.),p (Spveg.Exo/Nat)	34.89	0.00	0.38	5.00	-10.30	0.46	0.767
<i>Chondrohierax uncinatus</i>	y (.),p (Pets)	32.36	0.00	0.18	5.00	-9.04	1.71	0.165
<i>Rupornis magnirostris</i>	y (.),p (Spveg.Exo/Nat)	81.97	0.00	0.50	5.00	-33.84	0.61	0.778
<b>Falconidae</b>								
<i>Falco ruficularis</i>	y (.),p (Spveg.Exo/Nat)	65.59	0.00	0.60	5.00	-25.65	1.5	0.178
<i>Micrastur semitorquatus</i>	y (.),p (Spveg.Exo/Nat)	30.96	0.00	0.70	5.00	-8.33	0.67	0.843
<b>Pandionidae</b>								
<i>Pandion haliaetus</i>	y (.),p (Null)	26.37	0.00	0.29	4.00	-7.85	1.99	0.144
<b>Strigidae</b>								
<i>Ciccaba virgata</i>	y (.),p (Day+Spveg.Exo/Nat)	128.40	0.00	0.94	6.00	-54.97	2.77	0.019
<i>Glaucidium brasilianum</i>	y (.),p (Null)	99.88	0.00	0.24	4.00	-44.60	0.83	0.578

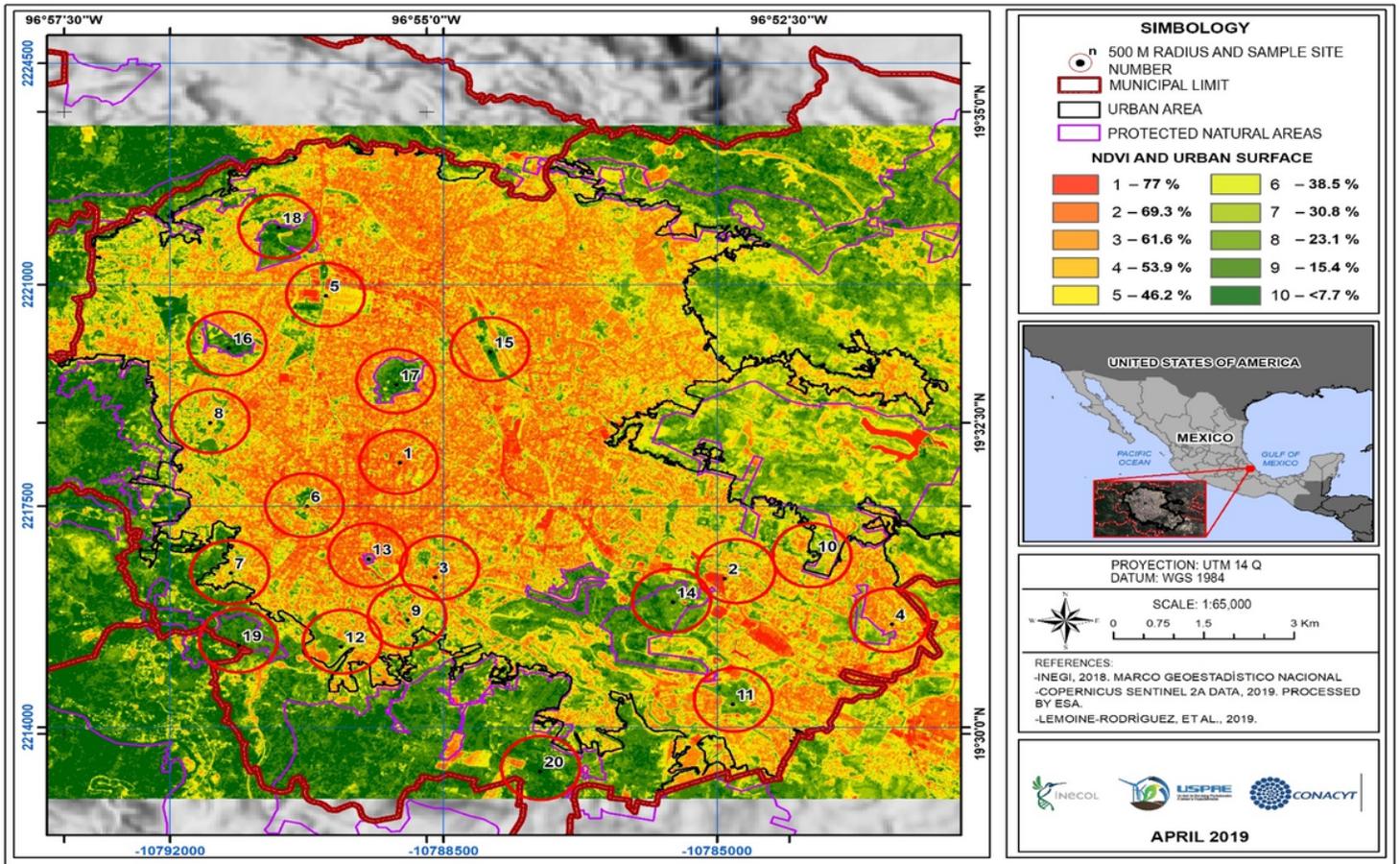
Detection variables (y ()): Spveg.Exo/Nat= Exotic/Native species proportion per site; Day= ordinal day sampled; Pets= Pets seen per site; Null= no detection variables used.  $\hat{c}$ = overdispersion parameter of the respective model. P-value=significance value from the overdispersion parameter.

Table 5. Occupancy models from analysis of detection data for selected species, classified by the Akaike information criterion for small samples (AICc). DAICc is the difference between the AICc value of the higher model. Each model includes the variable of the best detection model (see Table 4).

Species	Models	AICc	DAICc	$w_i$	k	LL	$\hat{c}$	P-value
<b>Accipitridae</b>								
<i>Accipiter cooperii</i>	y (HaUrb),p (S.E/S.N)	77.75	0.00	0.19	6.00	-27.19	0.58	0.809
<i>Buteo plagiatus</i>	y (Null),p (Null)	46.04	0.00	0.28	4.00	-17.69	1.07	0.452
<i>Buteo platypterus</i>	y (Null),p (S.E/S.N)	34.89	0.00	0.58	5.00	-10.30	0.44	0.81
<i>Chondrohierax uncinatus</i>	y (Null),p (P)	32.36	0.00	0.58	5.00	-9.04	1.65	0.169
<i>Rupornis magnirostris</i>	y (HaUrb),p (S.E/S.N)	77.71	0.00	0.19	7.00	-27.19	0.59	0.807
<b>Falconidae</b>								
<i>Falco rufigularis</i>	y (Null),p (S.E/S.N)	65.59	0.00	0.39	5.00	-25.65	1.58	0.132
<i>Micrastur semitorquatus</i>	y (Null),p (S.E/S.N)	31.01	0.00	0.58	5.00	-8.36	0.7	0.79
<b>Pandionidae</b>								
<i>Pandion haliaetus</i>	y (Null),p (Null)	26.37	0.00	0.49	4.00	-7.85	2.21	0.124
<b>Strigidae</b>								
<i>Ciccaba virgata</i>	y (Null),p (D+S.E/S.N)	128.40	0.00	0.34	6.00	-54.97	2.93	0.006
<i>Glaucidium brasilianum</i>	y (Null),p (Null)	99.88	0.00	0.24	4.00	-44.60	0.81	0.596

Occupancy variables (y(.)):HaUrb= Urban surface per site; Null= No occupancy variables used. Detection variables from best detection models (p(x)): S.E/S.N= Exotic/native species proportion per site; D= ordinal Day; P= pets; Null= no detection variables used.  $\hat{c}$ = overdispersion parameter of the respective model. P-value=significance value from the overdispersion parameter.

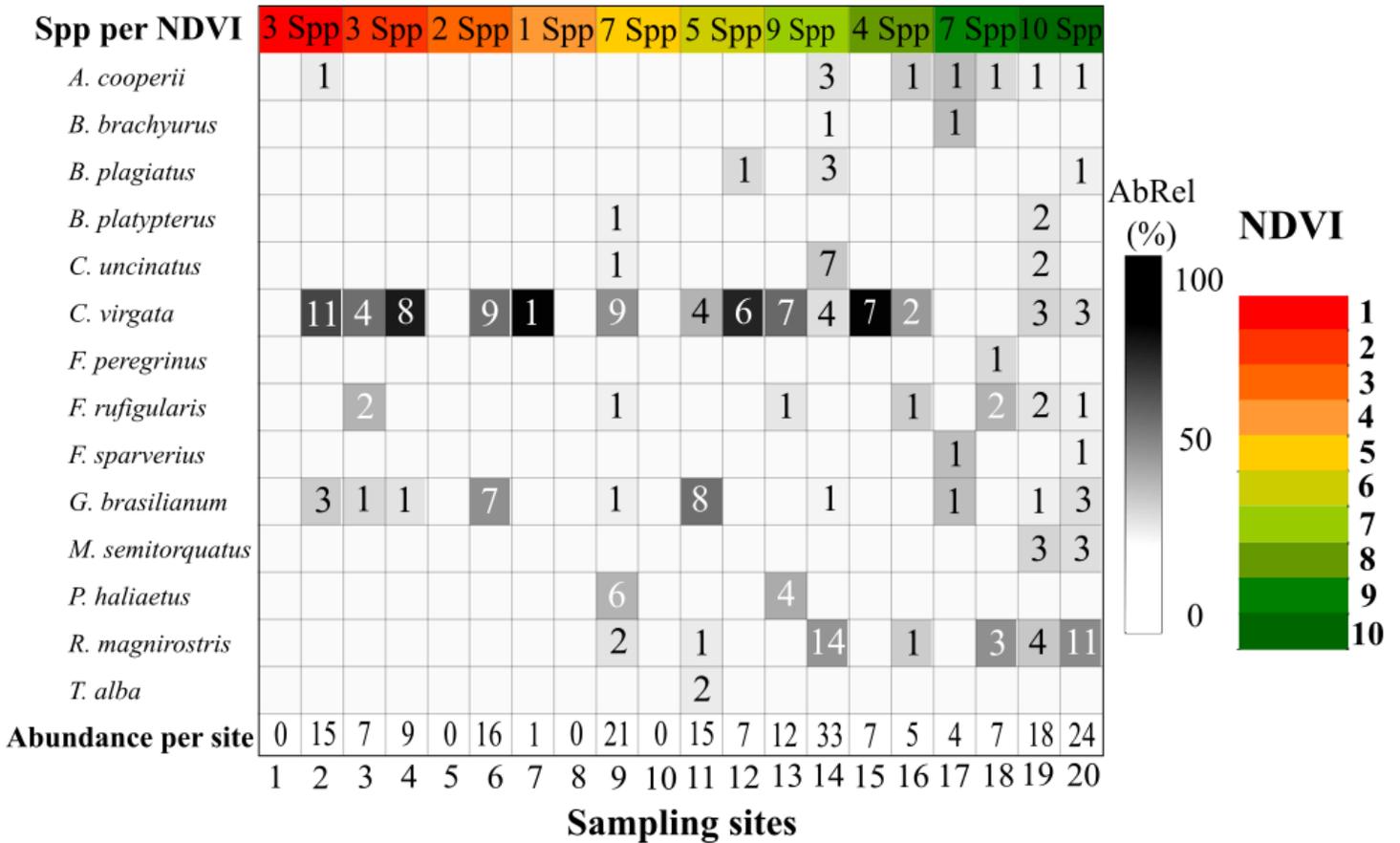
## Figures



**Figure 1**

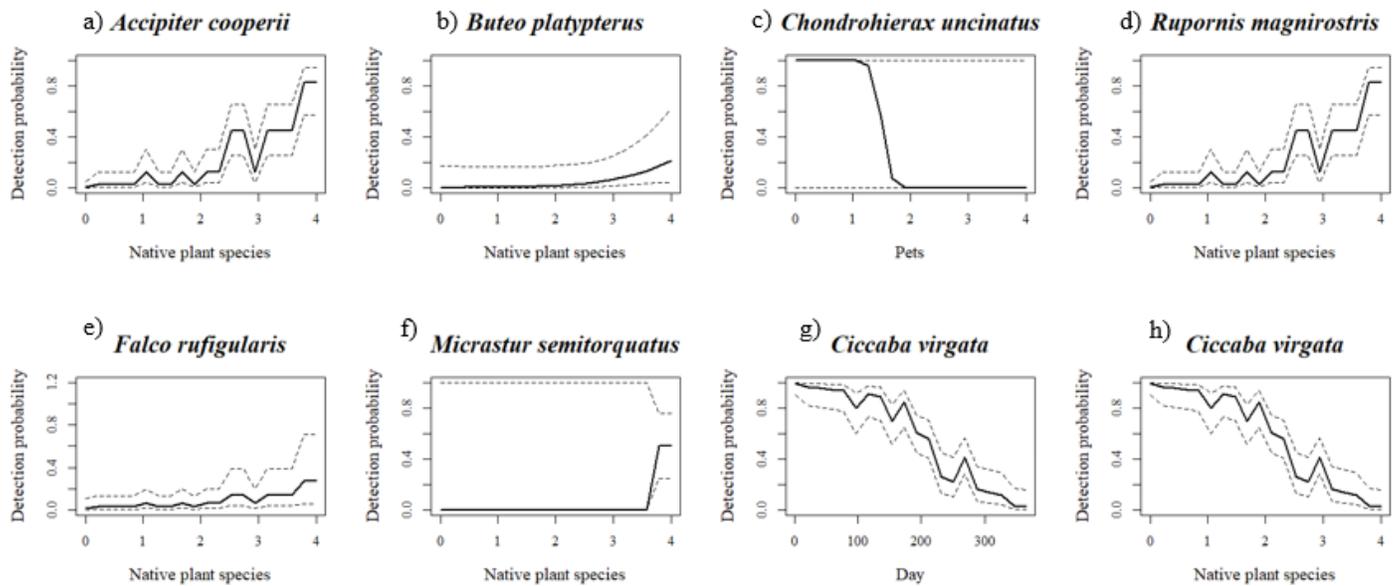
Sample sites (20) distributed along the urban gradient in Xalapa. Each site includes 500 m radius (in red) used for sampling and patch composition analysis. Urban gradient categories indicate: 1: 77% urban cover (red). 10: <7.7% urban cover (green)

## Relative abundance per site and species richness per urban gradient category



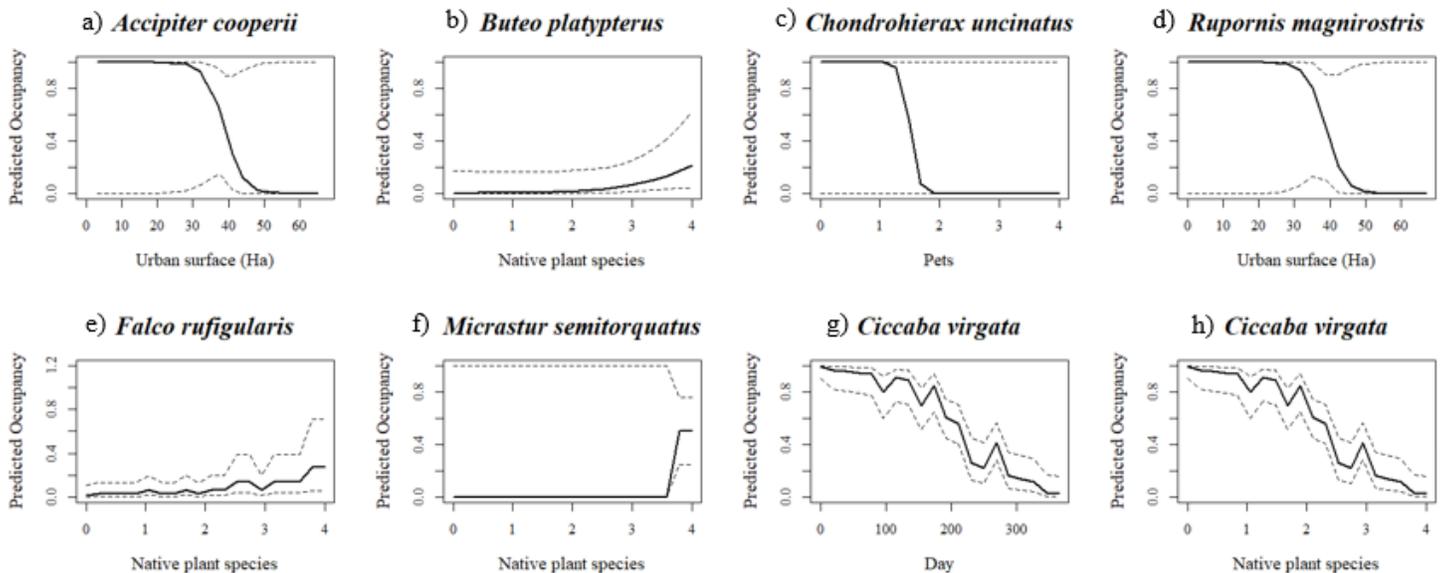
**Figure 2**

Abundance and species richness observed along the urbanization gradient in Xalapa. The species order is alphabetical. In the top (Spp per NDVI) is the species richness per urban gradient categories (NDVI; where 1= urbanized; 10= greenspaces). The numbers inside show the abundance for each species per site, and on the bottom (Abundance per site) is the summary of birds observed per site. Color boxes in inside indicate the percentage of relative abundance for each species per site



**Figure 3**

Detection probability from top detection models for 7 raptor species in an urban gradient in Xalapa (see table 4). For *A. cooperii*, *B. platypterus*, *R. magnirostris*, *F. ruficularis*, *M. semitorquatus* best detection models includes Native and exotic plant species proportion (0= no native species/many exotic species; 4: many native species/no exotic species). Best detection model for *C. uncinatus* includes pets (0=no pets present; 4=many pets present); Best detection model for *C. virgata* includes sampling day in ordinal number (Day), and Native/exotic plant species proportion



**Figure 4**

Predicted occupancy from top detection models for 7 raptor species in an urban gradient in Xalapa (see table 5). Predicted occupancy for *A. cooperii*, and *R. magnirostris* includes urban surface and the best detection model (Native and exotic plant species proportion). For *B. platypterus*, *C. uncinatus*, *F. ruficularis*, *M.*

semitorquatus and *C. virgata* no occupancy variables were used (Null model), the best predictors were the detection models only in table 4