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WHAT MATTERS MOST? RELATIVE EFFECT OF URBAN HABITAT TRAITS AND HAZARDS ON URBAN PARK BIRDS

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Resumen. – ¿Qué importa más? Efecto relativo de la actividad humana, la infraestructura urbana, las características de la vegetación y la presencia/abundancia del gorrión común (*Passer domesticus*) sobre las comunidades de aves de parques urbanos. – Debido a los cambios que causa en la estructura de los hábitats, la urbanización representa una amenaza para la biodiversidad. Sin embargo, dichos efectos dependen de la capacidad de las especies de vida silvestre para tolerar las amenazas y aprovechar los recursos urbanos. En este trabajo medimos la importancia relativa que tienen las actividades humanas, la presencia/abundancia de depredadores potenciales de aves, la infraestructura urbana, las características de la vegetación y la presencia/abundancia del gorrión común sobre once especies de aves que habitan en parques urbanos de la ciudad de México. Nuestros resultados muestran que las variables más importantes que explican la abundancia de las especies de aves estudiadas son las características de la vegetación, seguidas por la infraestructura urbana y el número de depredadores potenciales. Los resultados de este trabajo muestran que la magnitud relativa del efecto que tiene el juego de variables estudiadas es especie-dependiente. Así, proponer actividades generalizables de manejo y planeación de parques urbanos podría ser contraproducente. Sin embargo, basados en nuestros resultados, sugerimos tres actividades de manejo que podrían aumentar el número de especies de aves nativas en parques urbanos: (1) incrementar la abundancia de árboles viejos en parques urbanos; (2) evitar la inclusión de componentes de infraestructura urbana en parques; y (3) elegir cuidadosamente especies de arbustos que podrían atraer un gran número de especies de aves. Aunque la presencia/abundancia del gorrión común y el número de transeúntes no representaron la variable principal que explicó la abundancia de ninguna de las especies de aves estudiadas, nuestros resultados sugieren que disminuir los valores de estas variables podría aumentar la abundancia de otras especies nativas en parques urbanos.

Abstract. – Through habitat trait changes, urbanization can represent a threat to biodiversity. However, such effects depend on the capacity of wildlife to tolerate urban-related hazards and use urban resources. In this study, we measured the relative magnitude of the effect that human activity, the presence/abundance of potential bird predators, urban infrastructure, vegetation characteristics, and the presence/abundance of House Sparrows have on native urban park bird species. Our results show that the most important variables explaining the abundance of the studied bird species were vegetation ones, followed by both urban infrastructure variables and the number of potential bird predators. The results of this study show that the relative magnitude of the studied set of variables is species-dependent. Thus, proposing generalized park management and planning activities based on this study could be misleading. However, based on our results, we suggest three management and planning activities that could enhance native bird species numbers within urban parks: (1) increasing the abundance of old trees in urban parks; (2) avoiding urban infrastructure components within parks; and (3) carefully choosing shrub species that could attract large number of birds. Although the presence/abundance of House Sparrows and the number of passing pedestrians were not the principal variable explaining the abundance of any of the studied species, our results suggest that lowering the abundance of House Sparrows and passing pedestrians could enhance the abundance of native bird species in urban parks. *Accepted 23 September 2010.*

Keywords: Bird communities, avian ecology, urban ecology, urbanization, Mexico City, Neotropics.

INTRODUCTION

Urbanization leads to major changes in habitat structure and composition that negatively affect bird communities (see Chace & Walsh 2006, Evans *et al.* 2009, MacGregor-Fors *et al.* 2009, and references therein). However, the effects that urbanization can have on birds depend on their capacity to tolerate urban-related hazards and their ability to use the array of resources found within urban systems (Emlen 1974, Shochat 2004). In fact, urban-dwelling birds have been classified in relation to their use of urban areas as: (1) suburban/urban-adaptable – species able to exploit urban resources at low-developed urban areas, and (2) urban-exploiters – species fully capable to exploit urban resources and reach their highest population densities within highly-developed urban areas (Blair 1996, McKinney 2002).

Within human settlements, urban green areas play an important ecological role for birds by offering suitable habitat within urban matrixes (Gavareski 1976, Lussenhop 1977, Jokimäki 1999, Fernández-Juricic & Jokimäki 2001, González-Oreja *et al.* 2007, Shwartz *et al.* 2008). In particular, urban parks have been identified as the urban land use that encompasses most rich and complex bird communities within cities (Jokimäki 1999, Sandström *et al.* 2006, Vallejo *et al.* 2009, Ortega-Alvarez & MacGregor-Fors 2009, Khera *et al.* 2009, MacGregor-Fors *et al.* in press, among others). Previous studies focused on the ecology of bird communities within urban green areas have concentrated their research on the effects that habitat traits, principally vegetation ones, have on them (Gavareski 1976, Jokimäki 1999, Sandström *et al.* 2006, Khera *et al.* 2009). Others have underlined the negative effects that human activities (e.g., moving vehicles, passing pedestrians) can have on urban-dwelling birds (Blair 1996, Ortega-Álvarez & MacGregor-Fors 2009). Also, a recently published study shows how the presence and abundance of an aggressive exotic urban-exploiter species (House Sparrow, *Passer domesticus*) distributed along the Americas (Kalinoski 1975, Gowaty 1984, Blair 1996, Ortega-Álvarez & MacGregor-Fors 2009) can have dramatic negative effects on bird communities in urban and agricultural areas (MacGregor-Fors *et al.* 2010). Nevertheless, the relative magnitude of the effects that these factors can have on urban-dwelling bird species remains unknown.

The aim of this study was to identify the type and relative magnitude of the effects that human activity, the presence/abundance of potential bird predators, urban infrastructure, vegetation characteristics, and the presence/abundance of House Sparrows have on native urban park bird species. We expected vegetation characteristics to play an important role on determining the abundance of the studied urban park species. Also, we expected House Sparrows to negatively affect similar-sized species due to higher competitive interaction between similar-sized species (Leyequién *et al.* 2007), and generalist birds due to high food-resource competition with other broad dietary species (Gavett & Wakeley 1986, Kimball 1997). Although we expected human activity, habitat characteristics, and the presence/abundance of House Sparrows to have an effect on native urban park birds, we expected strong and positive effects regarding vegetation characteristics and strong negative effects caused by the presence/abundance of House Sparrows.

METHODS

Study area and bird surveys. This study was performed in the Metropolitan area of Mexico City (referred as Mexico City hereafter). Mexico City is one of the most populated urban areas in the world (United Nations 2008), covers an area of > 1000 km², has a human population that surpasses 20 million inhabitants (Grimm *et al.* 2008), and exhibits an annual population growth of 0.8% (INEGI 2006). Although the establishment and continuous growth of this city has negatively affected wildlife within its urban area and surrounding systems, it still includes considerable biodiversity values (Nocedal 1987, Flores-Villela & Geréz 1994, Peterson & Navarro 2006).

We surveyed resident landbirds in five urban parks during summer (June to August) 2008 using 10 min unlimited radius point counts (Ralph *et al.* 1996). All birds seen or heard were recorded. Point counts were located at a minimum distance of 200 m from each other to assure data independence (Huff *et al.* 2000). We surveyed 30 point count repetitions at each park: (1) Reserva de Cerro del Judío; (2) Parque Ecológico La Loma; (3) Parque Ecológico Las Águilas - Japón; (4) Viveros de Coyoacán, and (5) Jardín Ramón López Velarde (referred as JUD, LOM, AG, COY, RLV, respectively hereafter). All parks are located within the 'intra-urban' area of Mexico City (*sensu* MacGregor-Fors 2010) and are surrounded by highly developed areas (> 80% built cover in a 1-km^2 scale as suggested by Marzluff *et al.* 2001 to categorize urban landscapes).

Habitat characterization. We measured 17 variables that describe urban traits (i.e., urban infrastructure, vegetation characteristics, park management) and hazards (i.e., human activity, presence/abundance of potential bird predators, presence/abundance of House Sparrows) at each sampling point: (1) cemented area (e.g., areas covered by small storage constructions, playgrounds, paved tracks); (2) number of electric light poles; (3) number of vegetation strata (i.e., tree, shrub, herbaceous plants); (4) tree species richness; (5) tree density; (6) tree diameter at breast height (DBH); (7) tree cover; (8) tree height; (9) shrub species richness; (10) shrub cover; (11) shrub height; (12) herbaceous plant cover; (13) herbaceous plant height; (14) number of passing pedestrians; (15) number of potential predators (i.e., dogs and cats); (16) presence/abundance of House Sparrows; and (17) park management. All cover and area variables were measured using a categorical classification with values ranging from 0–5, with 0 = 0% cover/area; 1 = 1–5% cover/area; 2 = 6–25% cover/area; 3 = 26–50% cover/area; 4 = 51–75% cover/area; and 5 = 76–100% cover/area. Park management was measured through a categorical classification ranging from 0–3, with values of 0 being non-managed parks, and values of 3 highly managed parks (e.g., herbaceous plant mowing, removal of fallen leaves, tree and shrub pruning, plant watering). All variables were measured in a 25 m area (1963.49 m^2). We measured both abiotic and biotic variables as they can affect the abundance and distribution of bird species (Heikkinen *et al.* 2007, Shwartz *et al.* 2008, Ortega-Álvarez & MacGregor-Fors 2009, MacGregor-Fors *et al.* in press).

Bird species selection. With the aim of selecting a group of bird species that allowed us to evaluate the relative effect that human activities, urban infrastructure, vegetation characteristics, and the presence/abundance of House Sparrows have on urban park birds, we used those species that were present in all five studied parks.

Data analysis. To identify the relative magnitude of the effects that human activity, the presence/abundance of potential bird predators, vegetation characteristics, urban infrastructure, and the presence/abundance of House Sparrows have on urban park birds, we performed regression trees using R (R Development Core Team 2010). Regression trees allow the interpretation of datasets where there are complex nonlinear relationships between the set of response and predictor variables (Deíath & Fabricius 2000). This analysis uses binary recursive partitioning to identify threshold values of a set of predictor variables, which can be a mix of continuous and categorical variables that are related to the response variable. Thus, regression trees identify successive critical values of predictor variables splitting the response variable in a dichotomous and hierarchical manner (Palomino & Carrascal 2007). These types of trees are analogous to multiple regression models, specifically those using forward selection of predictor variables (Crawley 2007). For this study, we performed one tree regression for each species using its abundance as response variable, and urban traits (i.e., urban infrastructure, vegetation characteristics, park manage-ment) and hazards (i.e., human activity, presence/abundance of potential bird predators, presence/abundance of House Sparrows) as predictor variables (as used for assessing other habitat-species associations; Ben-Shahar & Skinner 1988, Potapov *et al.* 2000, Lehmann *et al.* 2003, Hasui *et al.* 2007, Palomino & Carrascal 2007; MacGregor-Fors *et al.* 2010). Although the ideal distance between point counts to assure independence is of 250 m (Ralph *et al.* 1996) and our point counts were located at 200 m from each other, we considered the average number of recorded birds at each point count as an independent value, as the maximum distance at which we recorded birds was 95 m (average distance = 32.4 m ± SD 17.9 m).

RESULTS

We recorded 45 resident landbird species. Only 11 of them were recorded in the five studied urban parks, and thus were included in our analyses: (1) Lesser Goldfinch (*Spinus psaltria* – SPPS; formerly *Carduelis psaltria*); (2) Inca Dove (*Columbina inca* – COIN); (3) Black-headed Grosbeak (*Pheucticus melanocephalus* – PHME); (4) Rufous-backed Robin (*Turdus rufopalliatus* – TURU); (5) American Robin (*Turdus migratorius* – TUMI); (6) Berylline Hummingbird (*Amazilia beryllina* – AMBE); (7) Bewick's Wren (*Thryomanes bewickii* – THBE); (8) Canyon Towhee (*Melozone fusca* – MEFU; formerly *Pipilo fuscus*); (9) Bushtit (*Psaltriparus minimus* – PSMI); (10) House Finch (*Carpodacus mexicanus* – CAME); and (11) Black-backed Oriole (*Icterus abeillei* – ICAB). This set of species include 45% granivores (i.e., SPPS, COIN, PHME, MEFU, CAME), 18% insectivores (i.e., THBE, PSMI), 18% frugivores (i.e., TURU, TUMI), 9% omnivores (i.e., ICAB), and 9% nectarivores (i.e., AMBE).

Regression trees revealed that all the studied variables, with the exception of the number of vegetation strata, were related to at least one of the 11 studied urban park species. However, such relationships and their magnitudes varied among species. Following the consideration that the most important variable in a regression tree is the one at the first dichotomy (Crawley 2007), vegetation traits showed to be the most important variable for eight of the studied species, while both urban infrastructure variables (i.e., cemented area, number of electric light poles) and the number of potential bird predators showed to be the most important variables for the remaining species (Figs 1 and 2).

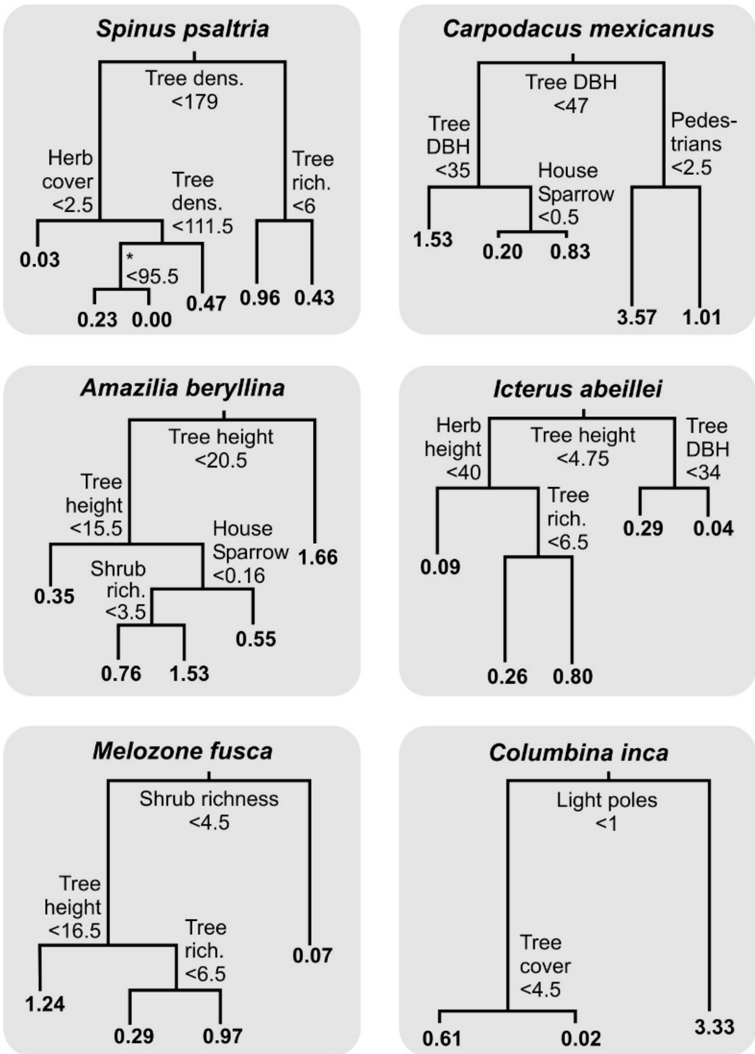

FIG. 1. Regression trees showing the relationship between urban traits and hazards and the abundance of SPPS, CAME, AMBE, ICAB, MEFU, and COIN. As regression trees use binary recursive partitioning to identify hierarchical threshold values of a set of predictor variables related to the response variable, they allow identifying scenarios under which a response variable changes in relation to the set of related predictor variables. * = tree density.

As stated in the methodological section, each regression tree indicates a scenario based on the specific thresholds for the related response variables. The interpretation of each tree relies on the dichotomic comparison of the abundance of each species according to the related variables. For example, the regression tree for SPPS showed to be related to three vegetation variables, as follows. When tree density was > 179, both high

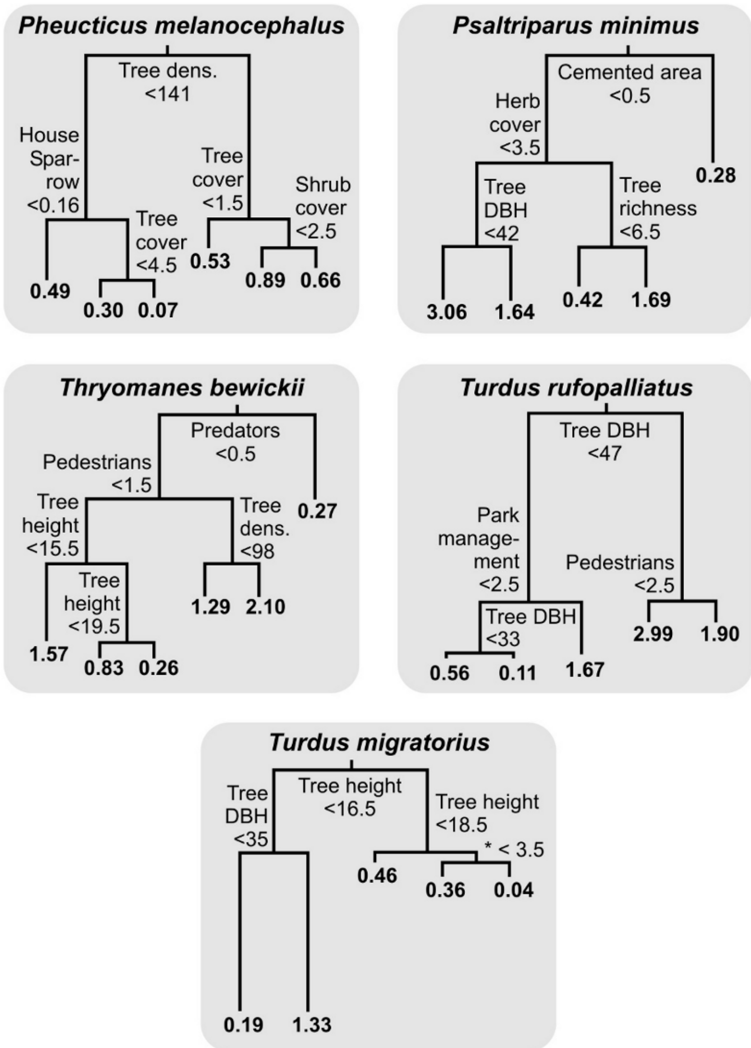

FIG. 2. Regression trees showing the relationship between urban traits and hazards and the abundance of PHME, PSMI, THBE, TURU, and TUMI. As regression trees use binary recursive partitioning to identify hierarchical threshold values of a set of predictor variables related to the response variable, they allow identifying scenarios under which a response variable changes in relation to the set of related predictor variables. * = shrub species richness.

and intermediate SPPS abundances were recorded, depending on the number of tree species present in the area. When tree density was < 179, herbaceous plant cover played an important role determining SPPS abundances, with values < 2.5 (~ 26.7%) related to very low SPPS abundances (0.03 average individuals per point count). When herbaceous plant cover was > 2.5 and tree density was > 111.5, SPPS abundances were higher (0.47 average individuals per point count), than when tree density was < 111.5 (0–0.23 average individuals per point count) (Fig. 1).

For each species, maximum and minimum average abundances were output by the regression tree models, representing those scenarios under which each species was recorded in highest and lowest abundances in relation to our surveys. SPPS maximum average abundance (0.96 individuals per point count) was recorded when tree density was > 179 and tree richness was < 6, while minimum values (0 individuals per point count) were recorded when tree density was < 179, herbaceous plant cover > 2.5, and tree density ranging from 95.5–111.5. CAME maximum average abundance (3.57 individuals per point count) was recorded when tree DBH was > 47 cm and < 2.5 pedestrians were recorded passing through the point count, while minimum values (0.20 individuals per point count) were recorded when tree DBH ranged between 35–47 cm and House Sparrows were absent (Fig. 1). AMBE maximum average abundance (1.66 individuals per point count) was recorded when tree height was > 20.6 m, while minimum values (0.35 individuals per point count) were recorded when tree height was < 15.5 m (Fig. 1). ICAB maximum average abundance (0.8 individuals per point count) was recorded when tree height was < 4.75 m, herbaceous plant height was > 40 cm, and tree species richness was > 6.5, while minimum values (0.04 individuals per point count) were recorded when tree height was > 4.75 m and tree DBH was > 34 cm (Fig. 1). MEFU maximum average abundance (1.24 individuals per point count) was recorded when shrub species richness was < 4.5 and tree height was < 16.5 m, while minimum values (0.07 individuals per point count) were recorded when shrub species richness was > 4.5 (Fig. 1). COIN maximum average abundance (3.33 individuals per point count) was recorded when > 1 electric light pole was present in the point count area, while minimum values (0.02 individuals per point count) were recorded when < 1 electric light pole was present and tree cover was > 4.5 (~ 75% cover; Fig. 1). PHME maximum average abundance (0.89 individuals per point count) was recorded when tree density was > 141, and shrub cover ranged between 1.5–2.5, while minimum values (0.07 individuals per point count) were recorded when tree density was < 141, House Sparrow abundance was > 0.16 and tree cover was > 4.5 (~ 75% cover; Fig. 2). PSMI maximum average abundance (3.06 individuals per point count) was recorded when cemented cover was < 0.5 (~ 1.25% cover), herbaceous plant cover was < 3.5 (~ 50 % cover), and tree DBH was < 42 cm, while minimum values (0.28 individuals per point count) were recorded when cemented cover was > 0.5 (~ 1.25 % cover; Fig. 2). THBE maximum average abundance (2.10 individuals per point count) was recorded when the number of potential bird predators was < 0.5, the number of passing pedestrians was > 1.5, and tree density was > 98. The regression tree for THBE revealed two scenarios under which lowest average abundance values were recorded (0.27 average individuals per point count): (1) when the number of potential bird predators was > 0.5; and (2) when the number of potential bird predators was < 0.5, the number of passing pedestrians was < 1.5, and tree height was > 19.5 m (Fig. 2). TUMI maximum average abundance (1.33 individuals per point count) was recorded when tree height was < 16.5 m and tree DBH > 35 cm, while minimum values (0.04 individuals per point count) were recorded when tree height was > 18.5 m and shrub richness was > 3.5 (Fig. 2). Finally, TURU maximum average abundance (2.99 individuals per point count) was recorded when tree DBH was > 47 cm and the number of passing pedestrians was < 2.5, while minimum values (0.11 individuals per point count) were recorded when tree DBH ranged between 33–47 cm and park management was < 2.5 (Fig. 2).

DISCUSSION

Habitat traits, including those concerning humans, often determine the presence of bird species in specific locations (Swift *et al.* 1984, Melles *et al.* 2003, Díaz *et al.* 2005, Ortega-Álvarez & MacGregor-Fors 2009). Our results show that all of the measured variables played an important role in determining the presence and abundance of the studied bird species, with the exception of the number of vegetation strata. It seems that the latter did not show any relationship due to the lack of variance, as 93% of the surveyed locations exhibited three vegetation strata. The most important variables explaining the abundance of the studied bird species were vegetation ones, followed by both urban infrastructure variables (i.e., cemented area, number of electric light poles) and the number of potential bird predators. Finding that vegetation traits play a crucial role in determining avian ecological processes was not surprising as several previous studies have found strong relationships between them and the abundance of urban-dwelling bird species (Gavareski 1976, Clergeau *et al.* 1998, Jokimäki 1999, McKinney 2002, Melles *et al.* 2003, MacGregor-Fors 2008, Shwartz *et al.* 2008, Ortega-Álvarez & MacGregor-Fors 2009, González-Oreja *et al.* in press). On the other hand, the number of light poles showed a positive relationship with the abundance of COIN, a species closely related to highly disturbed areas, including highly developed human settlements (Mueller 2004), while the number of potential bird predators and cemented area exhibited negative relationships with the abundance of two bird species (i.e., THBE, PSMI) related to semi-open shrubby habitats with considerable tree cover (Sloane 2001, Kennedy & White 1997). The latter indicates that those species that are tightly related to human disturbance are positively related to habitat traits that negatively affect other species, while more sensitive species are susceptible to urban development and potential predator abundance (Mitchell & Beck 1992, Coleman *et al.* 1997, López-Flores *et al.* 2009, MacGregor-Fors *et al.* in press). As the results found in this study are species-dependent, we discuss the reported relationships based to the natural history of each studied bird species.

The variable that showed relationships with a higher number of species (n = 5; CAME, ICAB, PSMI, TUMI, TURU) was tree DBH, which is closely associated to the age of trees (and differs from tree height within urban areas due to pruning activities). Regression trees revealed that tree DBH was the most important variable determining the abundance of two bird species: TURU and CAME. TURU is closely related to open areas with large trees (Percevia Field Guides 2007), which in this study represent DBH values > 47 cm, while CAME tends to avoid areas without trees due to scarcity of perching and nesting structures (Hill 1993). Although tree DHB was not the most important variable determining the abundance of ICAB, PSMI, or TUMI, our results show interesting patterns related to the interaction between variables. For example, TUMI showed a positive relationship with tree DBH values > 35 cm when tree height was < 16.5 m, suggesting that when trees are heavily pruned, TUMI prefers sites with older trees. Also, ICAB showed a negative relationship with tree DBH when trees were higher than 4.75 m. This result suggests that ICAB prefers sites with tall slim trees.

Followed by tree DBH, tree species richness and height were related to four species. On the one hand, tree species richness was related to four species (i.e., ICAB, MEFU, PSMI, SPPS). As discussed above, ICAB and PSMI were related in this study to tree DBH and are closely related to tree forests. However, there is no other evidence that agrees with our finding that indicates a positive relationship between the abundance of MEFU and tree species richness. Although this species nests primary on trees and uses small to medium-sized trees as shelter (Johnson & Haight 1996), there seems to be resources that a larger number of tree species, other than feeding ones (as basically feeds on herbaceous plant seeds), that benefit MEFU; however this hypothesis remains to be tested. Finally, the relationship between the abundance of SPPS and tree species richness is different to the idea that this species tends to avoid dense forests (Watt & Willoughby 1999), which is discussed below.

Tree height was the most important variable explaining the abundance of TUMI and AMBE. As discussed above, our results suggest that TUMI prefers sites with wider trees (higher tree DBH) when their height is < 16.5 m, suggesting that when trees are pruned, TUMI prefers sites with older trees. AMBE showed a positive relationship with tree height, particularly in sites where trees were higher than 20.5 m having highest AMBE abundances (1.66 average individuals per point count). The latter seems to be explained by the close relationship that this hummingbird has with heavily forested areas, avoiding disturbed areas other than clearings with nectar-rich patches (Percevia Field Guides 2007). Although tree height was not the most important variable determining the abundance of MEFU and THBE, it revealed two differential effects on these birds. First, the abundance of MEFU was negatively affected by tree height when shrub richness was < 4.5, showing that MEFU prefers open areas with young trees to forage, as reported previously (Johnson & Haight 1996). Second, THBE showed higher abundances when trees were lower than 15.5 m in the absence or low abundance of predators and pedestrians. The latter agrees with previous studies that have identified that this species is related to open woodlands away from humans, with cat predation being an important determinant of their populations (Kennedy & White 1997).

Of the rest of the considered variables, three of them (i.e., tree density, number of passing pedestrians, presence/abundance of House Sparrows) were related to the abundance of three of the studied bird species. Tree density was the most important variable explaining the abundance of SPPS and PHME. As discussed above, although Watt & Willoughby (1999) reported that SPPS tend to avoid dense forests, our results show that larger number of SPPS are present in areas where tree density is > 179. However, the regression tree for SPPS shows that when herbaceous plant cover is > 2.5 (~ 38 %), tree densities can have both positive and negative effects on the abundance of SPPS. As for PHME, their abundances were highest in the regression tree when tree density was > 141 and shrub cover value ranged between 1.5 and 2.5 (~ 9–26.7%), which represent well vegetated urban areas with understory, as reported by Ortega & Hill (2010). Although tree density was not the most important variable determining the abundance of THBE in this study, the regression tree revealed that summed to the importance of predators and pedestrian activity for this species, areas with tree density > 98 can increase its abundance ~ 26.75%.

Although the number of passing pedestrians was not the most important variable explaining the abundance of any of the studied bird species, it was related to the abundance of CAME, THBE, and TURU. Specifically, the number of passing pedestrians showed a negative relationship with the abundance of CAME and TURU, while exhibited both positive and negative relationships with the abundance of THBE. According to Fernández-Juricic & Tellería's (2000) study, pedestrians often influence the feeding activities of birds negatively. This could be the case of the recorded average abundance of CAME, which was more than three times higher when the number of pedestrians was < 2.5 and tree DBH was > 47 cm. However, THBE had highest numbers when the number of potential predators were < 0.5 and tree density was > 98, regardless that the number of passing pedestrians was > 1.5. In the case of TURU, when tree DBH was > 47, sites with < 2.5 passing pedestrians were related to a ~ 33 % increase of the average abundance of TURU per point count.

Although the presence/abundance of House Sparrows was not the most important variable explaining the abundance of any of the studied bird species, it was related to scenarios in which low abundances were recorded for three species (i.e., AMBE, CAME, PHME) ranging from 40 % to three times lower average abundance per point count. The relationship between the presence/abundance of House Sparrows and the average abundance of AMBE, CAME, and PHME were related to lower tree trait value scenarios (tree height < 20.5 m, tree DBH < 47 cm, and tree density < 141, respectively). The latter was not surprising, as the House Sparrow in Mexico is positively related to urban infrastructure and negatively related to vegetation traits (MacGregor-Fors *et al.* 2010, in press). This result confirms the negative effect that House Sparrows can have on urban-dwelling birds. House Sparrows often consume nectar in urban areas (Stidolph 1974, Leveau 2008), which could result in competitive interactions with other strict and facultative nectarivores, such as AMBE. Moreover, as House Sparrows feed mainly on grains when available, it could compete over food resources with other granivore and omnivore species, such as CAME and PHME (Lowther & Cink 2006).

While the rest of the studied variables were only related to one or two of the studied bird species, cemented area, the number of electric light poles, and the number of potential bird predators were the most important variables that explained the abundance of PSMI, COIN, and THBE, respectively. As discussed above, THBE showed higher abundances in open wooded areas with low human activities, with potential bird predators being the most important variable explaining its abundance, as reported by Kennedy & White (1997) for cats. The average abundance of COIN per point count was basically explained by the presence/absence of light poles, with an important negative influence of tree cover. Although the positive relationship between the number of electric light poles and the abundance of COIN seems odd, it could be based on two non-exclusive explanations: (1) COIN often perches in large groups (up to 5 individuals recorded in this study) using electric light cables for perching; and (2) the presence of electric light poles is negatively related to the presence of vegetation due to security matters, and thus their presence is also related to open areas, where COIN generally forages (Mueller 2004). Summed to the latter, finding that higher abundances of COIN exist in the absence of tree cover is not surprising, as the most common foraging sites for this species are open areas, where it consumes mostly grass seeds (Mueller 2004). Also, as suggested by Emlen (1974), electric light poles represent well distributed perching resources, from which a wider view of potential hazards can be achieved. Finally, cemented area played the most important role in determining the abundance of PSMI. Sites with cemented areas > 0.5 (~ 1.25%) showed to have lowest PSMI average abundance per point count (0.28), while varied from 0.42–3.06 under scenarios dependent of three vegetation traits (i.e., herbaceous plant cover, tree DBH, and tree species richness). Although PSMI can be found in a great array of habitats, ranging from temperate forests to arid shrublands, it is adapting to suburban and edge habitats (Sloane 2001). Thus, our results suggest that the density of urban constructions, often related to lesser vegetation components, can have a negative impact on the abundance of PSMI in urban habitats. However, a previous study shows that this species is present along the urban gradient (Ortega-Álvarez & Mac-Gregor-Fors 2009), nevertheless its numbers are lower as urban development increases (pers. observ.).

The general results of this study show that the relative magnitude of the studied set of variables is species-dependent, as recorded in previous studies (Jokimäki 1999, Morneau *et al.* 1999, Fernández-Juricic *et al.* 2001, Blumstein *et al.* 2005). Species-dependent responses have been previously associated to food habits (e.g., availability of food resources, suitable foraging micro-habitat; Young *et al.* 2007), breeding demands (e.g., availability of suitable nesting sites; Blair & Johnson 2008), self-maintaining requirements (e.g., availability of roosting and sunbathing sites; Emlen 1974), adaptation to anthropogenic disturbance (McClure 1989), and/or tolerance to the presence of aggressive invading competitors (i.e., House Sparrows; Kalinoski 1975, MacGregor-Fors *et al.* 2010). Thus, proposing generalized park management and planning activities based on our results could be misleading. However, based on the principal trait that determined the abundance of the studied species, we suggest three management and planning activities that could enhance native bird species numbers within urban parks. First, as higher DBH, height, and tree density tended to increase the numbers of some bird species pertaining to widely different feeding guilds (i.e., TUMI, CAME, TURU, AMBE, SPPS, PHME), urban parks with high abundance of old trees could attract a higher number of individuals pertaining to a large group of species adaptable to urban park conditions, as recorded by other authors for entire bird communities in the past (Munyenyembe *et al.* 1989, MacGregor-Fors 2008). Second, the presence of cemented cover and potential bird predators tended to reduce the abundance of two species (i.e., PSMI, and THBE), while the number of electric light poles was related to increases in the abundance of COIN. The latter suggests that urban infrastructure tends to benefit some species closely related to human disturbance and to negatively affect other species that are sensitive to urban development, as has been recorded in other urban systems (Blair 1996, Crooks *et al.* 2004, Chace & Walsh 2006, MacGregor-Fors *et al.* in press). Also, potential bird predators could have important negative effects on the abundance of native bird species (Coleman *et al.* 1997). Thus, avoiding off-leash dogs and controlling free-ranging and feral cats could positively affect the presence and abundance of native birds in urban parks. Third, shrub species richness, as most important explanatory variable, only showed a negative relationship with MEFU. However, this variable showed a positive relationship with the abundance of AMBE, probably because the shrub species present in the studied areas have energy-rich flowers. Although the latter could seem contradictory, previous studies have found both positive and negative relationships between shrubs and urban-dwelling birds (Jokimäki 1999, Fernández-Juricic *et al.* 2001, Melles *et al.* 2003, Leston & Rodewald 2006, Rodewald 2009, Rodewald *et al.* 2009). Thus, we suggest that park managers should carefully identify which shrub species could attract higher numbers of native bird species, as including shrubs in large areas could reduce House Sparrow abundances (MacGregor-Fors *et al.* 2010). Finally, although the presence/abundance of House Sparrows and the number of passing pedestrians were not the principal variable explaining the abundance of any of the studied species, our results suggest that lowering the abundance of House Sparrows and passing pedestrians could enhance the abundance of native bird species in urban parks.

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