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Are invasive House Sparrows a nuisance for native avifauna when scarce?

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Abstract

Biological invasions are the second most important cause of species extinction. Aided by processes such as transportation and urbanization, exotic species can establish and spread to new locations, causing changes in the function and structure of ecosystems. The House Sparrow is a widespread and highly abundant landbird associated to human presence. Previous studies performed in urban landscapes have suggested that this species could be acting, in synergy with urbanization, as a potential threat to native urban avian assemblages. In this study we assessed the relationship between House Sparrow density and native bird species richness in a region where the sparrows are scarce and sparsely distributed. We surveyed bird assemblages in and around four small-sized human settlements, considering three conditions in relation to House Sparrow presence: urban invaded, urban non-invaded, and non-urban non-invaded. To assess the potential detrimental role of House Sparrows on native bird species richness, we measured, additionally to sparrow densities, 20 predictor variables that describe vegetation structure and complexity, as well as urban infrastructure and human activities across four seasons of 1 year. Our results show that maximum shrub height was positively related to bird species richness, built cover was negatively associated with it, and House Sparrow invaded sites were related to a significant decrease of bird species richness, with increasing richness loss when more sparrows were present. Thus, we here provide evidence that urban areas can act in synergy with the presence of House Sparrows (even in low densities) in the urban-related species richness decline pattern.

Keywords Biological invasions · Bird density · *Passer domesticus* · Species composition · Species richness · Urban ecology

Introduction

Biological invasions are considered one of the main drivers of species extinctions, altering species richness and composition of native communities at different spatiotemporal scales (Bellard et al. 2016). When the individuals of exotic species establish and colonize new locations, successful biological invasions occur (Blackburn et al. 2011) and may alter local environmental processes and the structure of local native communities (e.g., nutrient cycles, trophic networks, fire and erosion regimes; Pyšek et al. 2012; Ricciardi et al. 2013; Simberloff et al. 2013). Although invasive birds are abundant across the globe (Blackburn et al. 2009), the magnitude and variability of their impact on native assemblages remains poorly understood (Kumschick and Nentwig 2010). It is notable that three avian species have been included in the list of 100 worst invasive alien species (Lowe et al. 2000; but see Kumschick et al. 2016): Common Myna (*Acridotheres tristis*), Red-vented Bulbul (*Pycnonotus cafer*; but see Thibault et al. 2018), and European Starling (*Sturnus vulgaris*). These three species alone have been responsible for massive damages to crops and infrastructure, but also for spreading diseases, and displacing native avifauna through predation and competition for nest cavities (Fisher and Wiebe 2005; Harper et al. 2005; Tindall et al. 2007; Grarock et al. 2012).

Cities are key components for avian invasions, not only as hubs for the deliberate trading of pets, but also by

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promoting the establishment and spread of diverse bird species in highly predictable systems (Vitousek et al. 1997; Sax and Brown 2000; Shochat 2004; Shochat et al. 2010). The filtering of regional avifaunas in urban settings generally results in depauperate avian assemblages, especially in heavily urbanized conditions, a niche that has been heavily exploited by generalists, often exotic and/or invasive species (Chace and Walsh 2006; Aronson et al. 2014; La Sorte et al. 2018). Given that many of these generalist urban exploiters are prone to experience population explosions in urban areas, they frequently dominate urban bird assemblages (Sol et al. 2014).

Urban invasive birds have been accounted for economic losses due to damages to buildings and other urban structures (Pimentel et al. 2001, 2005; Booy et al. 2017), as well as the spread of diseases on a global scale (Pedersen et al. 2006). However, there is a lack of agreement on the ecological impacts that invasive birds pose on native species (Linz et al. 2007; Strubbe and Matthysen 2007; MacGregor-Fors et al. 2010, 2011; Mori et al. 2017; González-Oreja et al. 2018; Luna et al. 2018). One of the most widespread urban-related invasive bird species is the House Sparrow (*Passer domesticus*), a species considered to be native to Eurasia and North Africa and that has been associated with humans for 10,000 years, since the appearance of agricultural practices (Anderson 2006; Sætre et al. 2012). This sparrow has been either intentionally or unintentionally introduced by humans in Australia, New Zealand, North America, South America, and South Africa (Anderson 2006). Regarding its North American invasion, it was successfully introduced to Northeastern United States in the 1850s and arrived to Mexico around the 1910s, establishing numerous and dense populations that expanded across the country in following decades, reaching Mexico City by 1930 (Wagner 1959). House Sparrow populations resulting from these invasion events have continued their range expansion southward to Central America (Anderson 2006).

House Sparrows are ecologically and physiologically plastic, with an extensive array of nesting habits (Kimball 1997; Nhlane 2000; Peach et al. 2008; Hoi et al. 2011), foraging behaviors, and dietary breadth (Guillory and Deshotels 1981; Kalmus 1984; Flux and Thompson 1986; Anderson 2006). Although its main food sources are seeds, it has an omnivorous diet in urban environments, ranging from nectar, fruits, insects, and even discarded human-food leftovers (Stidolph 1974; Gavett and Wakeley 1986; Clergeau 1990; Moulton and Ferris 1991; Leveau 2008; MacGregor-Fors et al. 2020). Behaviorally, the House Sparrow is aggressive with both its conspecifics and heterospecifics, often competing for nesting cavities and food resources (Kalinoski 1975; Gowaty 1984; Radunzel et al. 1997; Anderson 2006). It is also known to be an important source of pathogens (Rappole and Hubálek 2003; e.g., avian pox and malaria, West Nile Virus; Anderson 2006; Delgado-V and French 2012). Albeit the undeniable success of House Sparrows in North America, population declines have been recorded in the past decades along urban-agricultural landscapes of Western Europe (Summers-Smith 2003).

Previous studies have shown negative relationships between the presence and abundance of House Sparrows and other native landbirds. For instance, in a Central Western Mexico medium-size city, avian assemblages dominated by House Sparrows had lower bird species richness (MacGregor-Fors et al. 2010). In another study performed in Mexico City, the abundance of some native bird species showed to be negatively related with the presence and abundance of House Sparrows (i.e., Berylline Hummingbird–*Amazilia beryllina*, Black-headed Grosbeak–*Pheucticus melanocephalus*), with lower average abundance per point count ranging from 40% to 300% decreases (Ortega-Álvarez and MacGregor-Fors 2010). Moreover, the abundance of rare native birds was negatively associated with sites used by House Sparrows for roosting and breeding, such as lamp poles in a west-central Mexican city (MacGregor-Fors and Schondube 2011). Yet, results of a recent study performed in urban greenspaces of three Mexican cities suggest that House Sparrows are not related with declines in native species richness (González-Oreja et al. 2018). Based on all of the above, we consider that there is enough correlative evidence to acknowledge that House Sparrows can represent a potential competitor able to displace native species (Schondube et al. 2009).

In this study we assessed the relationship between House Sparrow density and native bird species richness in scenarios where sparrows are scarce and sparsely distributed. It is notable that these conditions, where House Sparrows are not hyper-abundant differ to those of previous studies focused on the potential effects to native avifauna, where sparrow densities are high (MacGregor-Fors et al. 2010; Ortega-Álvarez and MacGregor-Fors 2010). Thus, we surveyed bird assemblages in and around four small-sized human settlements in Central Veracruz (Mexico), where House Sparrows are present in low numbers, considering three different conditions: urban invaded, urban non-invaded, and non-urban non-invaded. Based on contrasting results related to the potential negative relationship between House Sparrows and native bird species richness, we tested the following hypotheses: (1) low densities of House Sparrows are associated with a lower bird species richness and composition, holding the pattern of previous studies evidencing the negative relationship regardless of sparrows' densities, and (2) low densities of House Sparrows do not relate to bird species richness nor its composition, and thus do not represent a nuisance for native avifauna when present in low densities.

Methods

Study area

We conducted this study in four human settlements from Central Veracruz: Xico, Teocelo, San Marcos de León, and Colonia Úrsulo Galván (referred to as Xico, Teocelo, San Marcos, and Úrsulo Galván hereafter; Table 1). The largest settlement in the region is Xico, with an extension of 2 km^2 and a population of ~18,650 inhabitants (INEGI 2010), followed by Teocelo (1 km^2, ~9950 inhabitants; INEGI 2010), San Marcos (0.7 km^2, ~7250 inhabitants; INEGI 2010), and Úrsulo Galván (0.14 km^2, ~1700 inhabitants; INEGI 2010). The studied settlements have similar urban infrastructure (mainly composed of one to two story houses, few commercial areas, few buildings with over four stories) and are embedded in a landscape with similar characteristics (hilly topography, presence of multiple water streams, and similar climate; INAFED 2010). It is notable that the study region was originally covered, in general, by tropical montane cloud forest, which has been partially replaced over the last century by shade coffee plantations, cattle ranches, and urban centers (Williams-Linera 2007; García-Franco et al. 2008).

Study design and field surveys

We followed a survey design that allowed us to assess the relationship between the presence and abundance of House Sparrows and native bird species richness, considering two dichotomies: (1) House Sparrow invaded / House Sparrow non-invaded sites and (2) built up environments (referred to as urban hereafter) / non-built sites (sensu MacGregor-Fors 2010). Given that House Sparrows are absent outside urban areas in the region, we considered three survey conditions: (1) urban House Sparrow invaded (UI), (2) urban House Sparrow non-invaded (UNI), and (3) non-urban House Sparrow non-invaded (NUNI). Due to differing sizes of the studied settlements and the presence and distribution of House Sparrows within them, our design was unbalanced, with a total of 110 survey sites (Table 1, Fig. 1).

MG-A performed 5-min point counts (25 m limited radius) from sunrise to 11:00 h, recording all birds seen or heard at each survey site in four seasons: spring, summer, fall, and winter (i.e., April 2016, July 2016, October 2016, January 2017). MG-A measured the exact distance from point-count locations to each recorded bird individual with a rangefinder (Bushnell Yardage Pro Sport 450). We established point counts at least 150 m apart from each other to be considered as independent sampling units (Ralph et al. 1996; Bibby et al. 2000; Huff et al. 2000).

Predictor variables

We measured 20 predictor variables within the same 25 m radius area in which birds were counted, once every surveyed season, to describe the environmental characteristics of each survey site. To describe vegetation structure and complexity, we recorded: (1) tree richness (morphospecies), (2) tree cover (%), (3) number of trees, (4) maximum tree height (m), (5) maximum diameter at breast height of trees (DBH) (cm), (6) shrub richness (morphospecies), (7) shrub cover (%), (8) maximum shrub height (m), (9) herbaceous plant richness (morphospecies), (10) herbaceous plant cover (%), and (11) maximum herbaceous plant height (m). To describe urban infrastructure and human activities, we recorded: (1) number of buildings, (2) maximum building height (m), (3) minimum building height (m), (4) number of light and electric poles, (5) number of cables, (6) number of windows, (7) passing cars per minute, and (8) number of pedestrians per minute. Additionally, we quantified built cover (%) in the 25 m radius survey area using satellite images from 2016 on Google Earth Pro (2018).

Data analysis

We computed the statistical expectation of species richness for each condition using rarefaction procedures with EstimateS, which allows statistical comparisons among treatments through the repeated re-sampling of all pooled samples based on their recorded abundances (Gotelli and Colwell 2001;

Table 1 Number and distribution of survey sites in the three conditions of the studied urban settlements

Study region	Settlement size (km^2)	UI[a]	UNI[b]	NUNI[c]	Latitude (N)	Longitude (W)	Elevation[d] (m a.s.l.)
Xico	2	9	12	21	19° 25′ 21.72”	97° 0′ 33.48”	1320
Teocelo	1	1	19	20	19° 23′ 7.08”	96° 58′ 30”	1160
San Marcos	0.7	3	7	10	19° 25′ 22.8”	96° 57′ 59.04”	1100
Úrsulo Galván	0.14	0	4	4	19° 25′ 45.84”	96° 58′ 41.88”	1140

[a] urban House Sparrow invaded, [b] urban House Sparrow non-invaded, [c] non-urban House Sparrow non-invaded, [d] Elevation was retrieved from INEGI (2010)

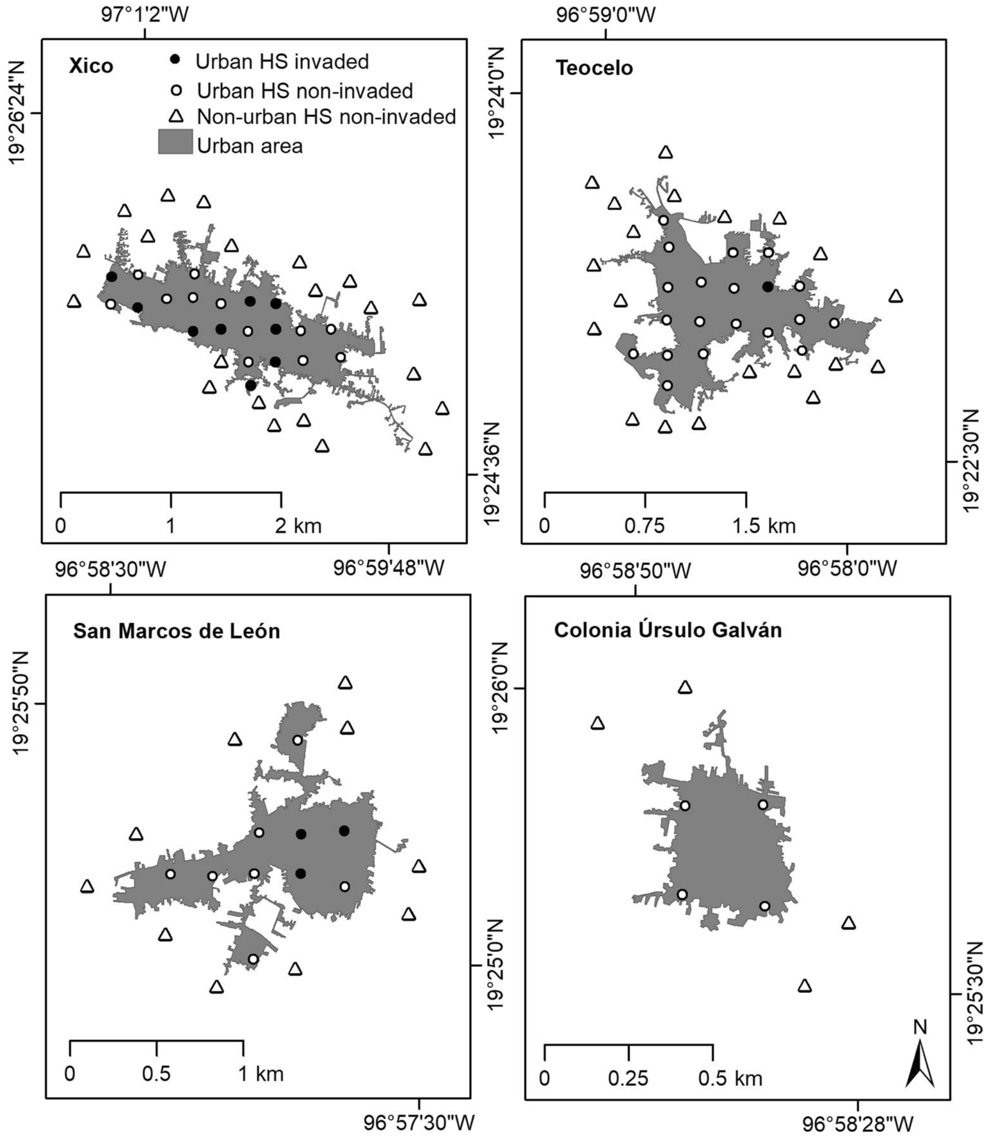

Fig. 1 Study areas and sampling locations. Map scales differ for graphical purposes. HS = House Sparrow

Colwell 2013). For comparisons among conditions we contrasted the 84% confidence intervals of the computed statistical expectations and considered statistical differences with $\alpha = 0.05$ when confidence intervals did not overlap (following MacGregor-Fors and Payton 2013). We used 84% confidence intervals as 95% confidence intervals fail to indicate statistical differences with $\alpha = 0.05$ (MacGregor-Fors and Payton 2013). Given that sampling effort varied among conditions, we used a factor of extrapolation of 2.5 for the smallest sample (i.e., UI) to robustly contrast its species richness calculations with the other two conditions at the same sampling effort (i.e., UNI, NUNI) (Gotelli and Colwell 2001; Colwell 2013).

We performed a multivariate Bray-Curtis cluster analysis (i.e., average linkage) using the package 'vegan' in R (Oksanen et al. 2016; R Development Core Team 2018) to describe similarities in bird assemblage composition among the studied conditions. Taking into account the 20 measured predictor variables and to avoid statistical issues related with multi-collinearity, we identified moderate-to-highly correlated variables (i.e., $r > 0.5$, $P < 0.05$), keeping those with highest variance. We used the remaining variables, including House Sparrow abundance per point count, in a generalized additive model (GAM) to explore their relationship with bird species richness. We used a GAM given that, as a variant of generalized linear models, additive models have different error structures and link functions able to provide a better fit for different types of variables, also allowing the use of non-parametric 'smoothers' (fitting procedure where the form of the curve is not predetermined but estimated through data; Wang 2014) to describe non-linear relationships (Crawley 2013). If House Sparrow abundances showed a significant relationship with species richness, we conducted a t-test to assess differences in built cover between sites with and without House Sparrow records.

To allow comparisons with results of previous studies in Mexico, we report the number of House Sparrows per point count, as well as estimated distance-corrected House Sparrow densities by season using Distance 6.2 (Thomas et al. 2010). Distance computes densities (ind/ha) based on the detection probability of individuals at increasing distances from the observer, as well as standardizing detection rates along concentric surveyed areas (Buckland et al. 2001).

Results

Over the course of four seasons (i.e., spring, summer, fall, winter) we recorded a total of 89 bird species of 29 families (Table S1 in Online Resource 1), of which 55% were recorded uniquely at the NUNI condition. In particular, we recorded 84 bird species at the NUNI condition, 36 at the UNI condition, and 20 at the UI condition. Nearly 25% of the recorded species are reported in the literature to be associated with well-vegetated areas, all of which we recorded at the NUNI condition, one of them also recorded at the UNI condition (i.e., Black-throated Green Warbler–*Setophaga virens*), and two at the UI condition (i.e., Magnolia Warbler–*Setophaga magnolia*, Rusty Sparrow–*Aimophila rufescens*) (Table S1 in Online Resource 1). Bird species richness at the UI condition was significantly lower when compared to that of the NUNI condition during almost all the year (summer, fall, winter) and compared to the UNI condition during summer (Table 2). Regarding species composition, the cluster analysis revealed that the UI condition shared less species with UNI and NUNI conditions, thus having a different assemblage composition across seasons (β = 0.13; Fig. 2).

Results of the GAM show that bird species richness was significantly related with season (Table 3). After taking into account the smoothing adjustment for the numerical variables (i.e., shrub richness, maximum shrub height, built cover, House Sparrow abundances, passing cars per minute), we identified that the relationship between maximum shrub height and bird species richness was positive (Fig. 3a), the one with built cover was negative (Fig. 3b), and the one with House Sparrow abundances showed three different scenarios

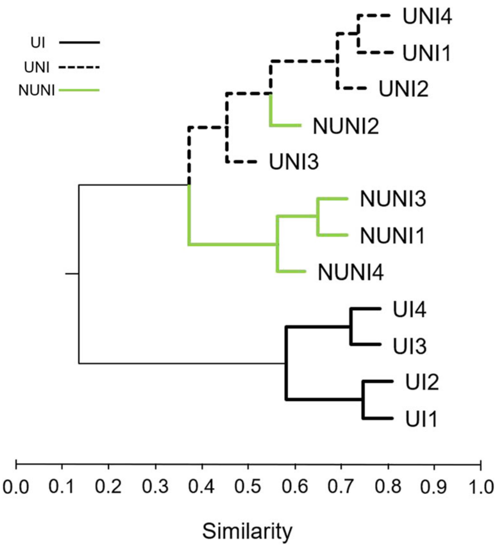

Fig. 2 Bray-Curtis group average link cluster showing avian assemblage composition patterns in the three studied conditions and seasons (UI = urban House Sparrow invaded; UNI = urban House Sparrow non-invaded; NUNI = non-urban House Sparrow non-invaded; numbers after study conditions represent seasons: 1 = spring, 2 = summer, 3 = fall, 4 = winter

(i.e., 0 individuals, 1–5 individuals, 6–12 individuals; Fig. 3c). Due to the complexity of the interpretation of such trichotomy, we calculated the statistical expectation of bird species richness for each scenario, finding a significant decrease in bird species richness as the number of House Sparrows increased (Fig. 3c). It is notable that we did not find differences for built cover values in sites with and without House Sparrow records ($t_{25} = -0.77$, $p = 0.45$; Fig. 4), showing that such decrease in species richness was not given by urbanization intensity.

The number of House Sparrows per point count was of 0.6 individuals during spring, 0.49 in summer, 0.45 in fall, and 0.4 in winter. Regarding distance-corrected densities, we recorded the highest House Sparrow density during winter (12.6 ind/ha 84% CI: 3.5–45.4), followed by spring (5.4 ind/ha 84% CI: 2.8–10.4), summer (2.5 ind/ha 84% CI: 1.2–4.8) and fall (2.3 ind/ha 84% CI: 1.0–5.0).

Table 2 Bird species richness (average ± 84% CI) across seasons in the surveyed conditions considering all studied settlements

	Season			
Condition	Spring	Summer	Fall	Winter
UI[a]	17.7 ± 4.3	10.7 ± 1.7	11.8 ± 4.4	14.5 ± 6.4
UNI[b]	19.9 ± 3.6	16.0 ± 3.5	12.4 ± 2.0	16.1 ± 3.2
NUNI[c]	25.5 ± 3.7	21.3 ± 3.5	25.6 ± 3.8	28.2 ± 3.6

[a] urban House Sparrow invaded, [b] urban House Sparrow non-invaded, [c] non-urban House Sparrow non-invaded

Table 3 GAM considering predictor variables describing vegetation characteristics and urban infrastructure in relation with native bird species richness

Variable	DF	χ^2	P
Season	3	24.03	<0.001
s (Built cover)	1	28.34	<0.001
s (Maximum shrub height)	1	9.13	0.002
s (House Sparrow abundances)	2	11.32	0.012
s (Shrub richness)	1	0.58	0.494
s (Passing cars per minute)	1	2.54	0.110

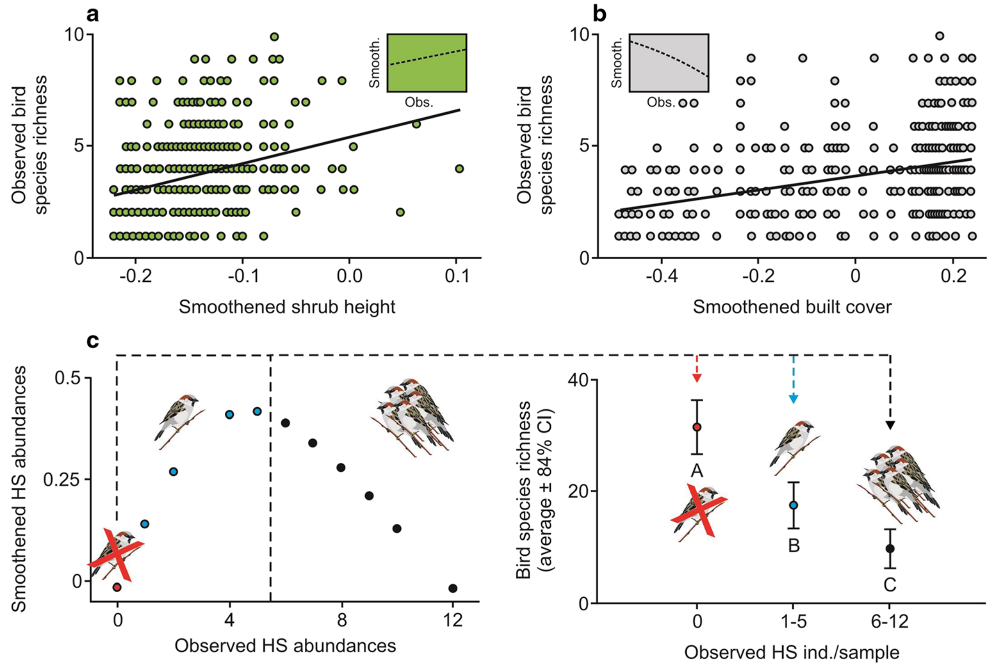

Fig. 3 In this graph we display variables that showed to be significantly related with bird species richness in the GAM. Panels **a** maximum shrub height and **b** built cover show the relationship with smoothened data, insets represent the best-fit for smoothened and observed values (positive for shrub height, negative for built cover). For **c** House Sparrow abundances, lower left panel corresponds to the best-fit adjustment, showing the three different scenarios of 0 individuals (red), 1–5 individuals (blue), and 6–12 individuals (black). Each scenario connects to its corresponding bird species richness in the lower right panel. Letters below the lower 84% CI bars stand for statistical significant differences. HS = House Sparrow

Discussion

The House Sparrow is a widespread and highly abundant landbird associated to humans (Aronson et al. 2014; Sol et al. 2014) that could be acting in synergy with urbanization as a potential threat to native avian assemblages, even when present in low numbers (MacGregor-Fors et al. 2010; Loss et al. 2015). Results of this study showed that vegetation elements are positively associated with bird species richness, meanwhile heavily urbanized areas are negatively related to it. Furthermore, sites with House Sparrows presence had lower bird species richness than non-invaded and non-urban areas. Also, the assemblages of invaded urbanized areas were more similar among themselves compared to those of non-invaded and non-urban areas. Altogether, our findings suggest the existence of different dynamics among bird species within urban areas where invasive sparrows are present, having an effect on both the number and composition of bird species.

Seasonality was related to an increase in bird species richness given by the amount of Neotropical-Nearctic migrants recorded in winter. It is noteworthy that our study area is located within one of the most important Neotropical-Nearctic bird migration routes (Ruelas-Inzunza et al. 2005). The positive relationship between maximum height of shrubs and bird species richness agrees with previous studies assessing avian ecology along urban-agricultural landscapes (Ortega-Álvarez and MacGregor-Fors 2009; Faggi and Caula 2017). This variable, as proxy of vegetation at each site, highlights the importance of structural stratification of vegetation for birds both in non-urban and urban areas (Cueto and de Casenave 1999; Napoletano et al. 2017). Built cover was negatively associated with bird species richness, which also agrees with previous studies assessing avian assemblages in cities (MacGregor-Fors and Schondube 2011; Luck et al. 2013; Schneider and Miller 2014; Faggi and Caula 2017). Actually, this relationship was not surprising, as urbanization has been directly linked to a decrease in bird species richness due to the loss of a wide variety of food resources, breeding sites, and additional factors inherent to urbanization (e.g., cat predation, window collision, parasitism; Santiago-Alarcon and Delgado-V 2017), among other causes (Emlen 1974; Chace and Walsh 2006).

Finally, House Sparrow numbers had a gradual negative effect on bird species richness, where sites having no sparrows (NUNI, UNI) showing significantly more bird species compared to sites with sparrows. Specifically, urban invaded areas (UI), with 1–5 House Sparrows had significantly more bird species than sites with 6–12 House Sparrows (Fig. 3). It is important to highlight that significant differences in bird species richness in sites where we recorded 1–5 and 6–12 House Sparrows were not related to built cover, as urbanized sites (invaded and non-invaded) had similar values (Fig. 4). Given that a possible confounding factor of the recorded relationship between House Sparrows and bird species richness could be

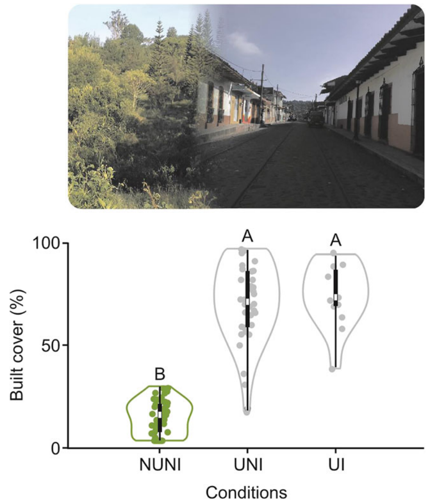

Fig. 4 Built cover at the studied conditions. Letters above error bars represent statistical differences

the potential association with the presence and abundance of other urban-related species (i.e., Great-tailed Grackle–*Quiscalus mexicanus*, Rock Pigeon–*Columba livia*, Tropical Kingbird–*Tyrannus melancholicus*), we assessed potential correlations between the presence and abundance of the most frequently recorded species with House Sparrows data. However, we found no significant or strong correlations between House Sparrow abundance and the abundance of other common urban-associated species ($r_S \leq |0.13|$, p-values <0.53; Table S2 in Online Resource 1). Therefore, our conclusion regarding the negative relationship between House Sparrows and native bird species richness holds true.

Altogether, our results add information to the scarce evidence that this invasive sparrow could be acting as a driver of native urban bird assemblages, even when present in low densities. It is important to note that House Sparrow numbers recorded in this study were much lower (i.e., 10–32 times lower in terms of relative abundance and 1.6–3 times lower in terms of density) than those reported in previous studies (i.e., ~20 ind/point count in MacGregor-Fors et al. 2010, 9.5–33.3 ind/ha in MacGregor-Fors et al. 2017; ~7 ind/point count in Ortega-Álvarez and MacGregor-Fors 2011a). Yet, similar low densities are reported for some of the native populations of the House Sparrow (Šálek et al. 2015), where this species is considered at risk (Summers-Smith 2003; BirdLife International 2004; Shaw et al. 2008).

Previous evidence has suggested that not only House Sparrows could represent a threat to similar sized and smaller granivore species through direct antagonistic interactions (Schondube et al. 2009), but also to species from other guilds and sizes, such as hummingbirds (Ortega-Álvarez and MacGregor-Fors 2010), as well as species with similar nesting habits (e.g., bluebirds, swallows; Kalinoski 1975; pers. obs.). House Sparrow presence along with the threats of urbanization (e.g., introduced predators, pollution, habitat destruction; Santiago-Alarcon and Delgado-V 2017) and indirect interactions (Marzal et al. 2011, e.g., parasite transmission to native birds via both invasive [novel weapon hypothesis] and migratory species; Marzal et al. 2018) can be driving the observed patterns. Thus, our results support that House Sparrows can act synergistically in relation with urbanization in the species richness decline pattern (Chace and Walsh 2006; Ortega-Álvarez and MacGregor-Fors 2011b, c; Aronson et al. 2014; Sol et al. 2014; MacGregor-Fors and García-Arroyo 2017).

We consider that further directions to test the effects of House Sparrows, in synergy with urbanization on native bird communities, require both laboratory and field experiments. In doing so, studies ought to consider balanced designs, taking into account diverse urban conditions (e.g., residential, industrial, commercial, greenspaces), including non-urban controls of different land uses (e.g., original vegetation, agricultural), and several House Sparrow abundance scenarios. Additionally, it is of the utmost importance to study House Sparrow intraspecific and interspecific interactions (e.g., feeding and nesting resources), as well as monitoring their populations in different spatiotemporal scales. Finally, and based on our field observations, we highlight the importance of the maintenance of vegetation cover and structure in urban areas, not only in large greenspaces but also in private gardens and along streets, with the aim of promoting the native avian assemblage diversity.

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