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# Vegetation regulations and upkeeping influence avifauna diversity within residential developments of Mexico City

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## Abstract

Biodiversity inclusive design is an emerging branch offering built-environment professionals the opportunity to support species conservation within urban landscapes through the design process and upkeep of those areas. Considering its recent emergence coupled with vegetation development time frames (i.e. life cycles), the evidence of how biodiversity inclusive design supports species composition is still lacking. Thus, past cases that inadvertently used biodiversity offer an opportunity to inform the potential trajectories of these frameworks and to identify the array of biodiversity enhancement actions available to designers looking to support species persistence within urban areas. Here, we assessed the avian community and vegetation characteristics of two housing developments within Zona Esmeralda. We discussed the ecological context to then understand the role of biodiversity inclusive practices as drivers of avifauna diversity. The ecological context was addressed by comparing our study with others that also addressed avifauna in urban areas within central Mexico. We found that Zona Esmeralda presented higher ecological value than what is traditionally expected from residential areas, which was attributed to habitat characteristics within our study, where greater vegetation complexity and availability were associated with higher bird diversity. Our study showed that vegetation upkeep (i.e. management/maintenance during construction and maintenance phases) was the strongest contributor to vegetation assemblages and therefore enhanced bird diversity. Our study highlights that even the partial use of biodiversity inclusive design principles (particularly relating to vegetation upkeep) can support biodiversity and boost the ecological value of such areas, when compared to areas that do not apply these principles.

**Keywords** Biodiversity inclusive design · Housing development · Vegetation complexity · Vegetation maintenance · Construction regulations · Species diversity

## Introduction

The biodiversity inclusive design approach is a branch emerging within urban landscapes, where members from built-environment disciplines (e.g. designers, planners, and urban ecologists) are coming together to minimise anthropogenic impacts associated with urbanisation and even restore native species (Hernandez-Santin et al. 2022). Some of the frameworks that use biodiversity inclusive thinking include animal-aided design (Weisser and Hauck 2017), biodiversity sensitive urban design (Garrard et al. 2018), and wildlife inclusive urban design (Apfelbeck et al. 2020). These frameworks share elements such as: (1) assessing species richness, composition, and distribution, as well as the factors that may influence them; (2) determining clear and measurable ecological targets for the project, including sought after ecosystem services or focal species that

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act as non-human stakeholders to design for, which may be used as indicators of success and/or may have ecological or socio-cultural importance; (3) designing to ensure availability of resources to meet ecological requirements of focal species; and (4) involving human communities in the design process and/or its upkeeping (i.e. management/maintenance jobs) to enable positive human-biodiversity interactions, foster connection to nature, and stewardship (Garrard et al. 2018; Hernandez-Santin et al. 2022).

While biodiversity inclusive design is a new approach to care for species in urban environments, ecosystem services have long been explicitly or implicitly recognised. Explicit recognition refers to the ecosystem services concept itself, arising in the 1980s (de Groot et al. 2017). By implicit recognition, we refer to the ancestral human-nature relationship. Implicit recognition suggests that aspects relating to ecosystem services and/or biodiversity inclusive design may arise in built environments even without a conscious intention, leading to the inclusion of biodiversity as an incidental stakeholder (Hernandez-Santin et al. 2023). For example, encroachment of housing into natural ecosystems or building parks within cities can occur in a pursuit of cultural ecosystem services, such as seeking aesthetic and/or recreational values, fostering nature's contribution to our cultural identities, and/or catering human's sense of place (Elands et al. 2019; National Trust 2017). Such aspects of biodiversity inclusive design may be unintentionally included as part of an area's design process or as biproduct of its management and/or upkeeping strategies. By upkeeping, we refer to maintenance works, such as mowing or irrigating green spaces.

Understanding cases with incidental biodiversity inclusive design is paramount to improve future (intentional) strategies. The earliest frameworks unequivocally relating to biodiversity inclusive design started to emerge in scientific literature between 2015 and 2020 (Hernandez-Santin et al. 2022). Therefore, there is still a lack of evidence of their ability to support species conservation due to the long periods required to develop vegetation assemblages. 'New' functional vegetation communities are expected to establish over decade-long time frames in natural and built environments alike, with high mortality rates of seedlings planted (Brunner and Cozens 2013; Gann et al. 2019; Hernandez-Santin et al. 2020; Parsons et al. 2023; Walker et al. 2007). For example, most oak species (*Quercus* spp.) reach reproductive maturity at ~20 years (Tantray et al. 2017). Distinctively, tree regeneration in urban environments occurs at lower rates than in natural environments, and is often an assisted endeavour that requires careful planning (Le Roux et al. 2014). Thus, past examples of incidental biodiversity inclusive design are still the most suited cases to inform the potential trajectories of these frameworks and to identify the array of biodiversity enhancement actions available to designers looking to support species survival within urban areas. Zona Esmeralda, located in the northwestern fringe of Mexico City, represents one of such examples.

One of the key requirements of biodiversity inclusive design involves the identification of focal species (single or group) to design for (Hernandez-Santin et al. 2022). Birds are considered bioindicators due to their high detectability and ubiquity, with specific species responding to a range of conditions depending on their traits (Morelli et al. 2021). In addition, avifauna represents one of the first precursors of biodiversity inclusive thinking (Fernández-Juricic and Jokimäki 2001; Lundberg et al. 2008). Thus, we chose the avifauna community as our focus group. Species assessments are key to understand the current state, as well as trends and trajectories of species compositional, structural, and functional attributes within ecosystems (McKenna et al. 2022).

Species' presence and abundance are shaped by temporal and spatial components that interact to drive patterns noticeable at different scales (Chave 2013). Seasonality is one of the main temporal drivers of avian communities, where patterns arise as migratory species flee harsh winters. For example, in North America, species migrate from northern parts of the subcontinent in autumn and reach southern subcontinental grounds (i.e. Mexico) to establish overwintering territories before migrating back around late winter or spring (Pacheco-Muñoz et al. 2022). Thus, the nonbreeding bird community in central Mexico is represented by migratory (stop overs or seasonal residents) and non-migratory (year-round residents) species, which could lead to greater diversity when hosting migratory species.

On the other hand, spatial components within an ecosystem are usually driven by the level of urbanisation, fragmentation, and habitat complexity (i.e. heterogeneity). Within the urbanisation gradient, areas closer to the urban core have lower biodiversity than their natural counterparts as bird diversity is negatively associated to human population size due to the greater number and levels of disturbances (e.g. anthropogenic activities, habitat loss, and fragmentation) (Gagné et al. 2016). Notably, urban ecosystems have homogeneous biodiversity, exhibit higher levels of fragmentation, and are less resilient with loss of ecological function and niches (McKinney 2006, 2008). Fragmentation in urban areas is ruled by island-biogeography principle, where impervious surfaces act as the 'ocean' and greenspaces act as 'islands' (Fernández-Juricic and Jokimäki 2001). While larger greenspaces will hold more species, the connectivity gradient suggests that areas with better connectivity to natural environments will hold greater species richness (Rastandeh et al. 2018). Along the habitat heterogeneity gradient, areas with greater vegetation complexity are often related to higher species diversity (Nielsen et al. 2014).

The goal of this study was to assess the avifauna community of two housing developments within Zona Esmeralda, in context of their incidental use of some principles of biodiversity inclusive design. For this, we assessed the ecological background to then interpret our results within a biodiversity inclusive design approach, using a literature review, avifauna surveys, and field-based and remote sensing vegetation monitoring. For the ecological context we assessed differences within our study area and when compared to previous studies in the region. We tested hypotheses relating to the potential drivers of species richness when comparing studies: sampling effort, seasonality, fragmentation gradient, urbanisation gradient, and heterogeneity gradient. Considering the temporal component, studies with greater sampling effort and monitoring over longer periods (covering more seasons) of time would be expected to hold greater species diversity. Considering the spatial component, higher diversity could be expected for studies monitoring larger parks with greater habitat complexity and located closer to natural areas, or those acting as a remnant themselves. For the biodiversity inclusive design context, we reviewed regulations and vegetation upkeep strategies within housing developments and related them to our findings for the habitat component.

## Study area

Zona Esmaralda is located in the municipality of Atizapan de Zaragoza, Estado de Mexico, in the northwestern portion of one of the largest urban encroachments in the Americas: Greater Mexico City. Zona Esmeralda is mostly residential and is considered relatively well preserved, as it also includes protected and unprotected remnants of oak forests (*Quercus* sp.), induced grasslands, and xeric scrublands (Capron and Esquivel Hernández 2016; Medina Lemus and Tejero-Díez 2006). The residential areas are composed of a series of gated communities, where common non-native vegetation includes *Eucalyptus* sp., *Bougainvillea* sp., and *Juniperus* sp. (personal observation). Considering that Zona Esmeralda started to develop around the late 1970s (Mauricio Hernandez, personal communication), an important portion of its non-native vegetation has progressed into functional developmental stages. This means that non-native vegetation has grown and is established, allowing species to use them to obtain feeding resources, shelter, etc. Notably, two of the authors (LHS and CHS) grew up close to and lived in Zona Esmeralda, allowing understanding of the historical context of this area.

We chose two housing developments as our study sites: Condado de Sayavedra (from hereon: Sayavedra) and Loma de Valle Escondido (from hereon: Loma) (Fig. 1). Sayavedra and Loma are separated by a linear distance of 1.24 km and 2.0 km at its shortest and longest distances, respectively. Sayavedra is the northern-most and largest housing development in Zona Esmeralda, covering an area of 478 ha and holds two intermittent creeks, natural patches of oak forest, and a large equestrian area (107 ha). Loma is five times smaller (91 ha), holding preserved oak forest mostly along a single intermittent creek, and is adjacent to a golf course on the southwestern portion of it.

## Methods

### Avifauna

Avifauna surveys conducted in Zona Esmeralda were complemented with data obtained from a literature review to perform a meta-analysis, providing ecological context of our study area. Given logistical constraints, including observer safety, we leveraged existing studies to adopt a comparative approach.

#### Literature review

We conducted a systematic literature review to select studies that focused on avifauna within the vicinity of our study area. Inclusion criteria were: research manuscripts, surveys monitoring the breeding community, reporting > 1 site, within a buffer of 150 km of our study area, and within the same bioregion. This buffer was chosen for proximity, due to spatial autocorrelation at the regional scale -island biogeography theory-, where spatially closer organisms or environmental conditions are expected to be more similar to each other than those that are further apart. Differentiating bioregions was important, as Nearctic and Neotropical birds are known to respond differently to urbanisation (González-Oreja 2011). Our study area is part of the 'transitional' bioregion (MacGregor-Fors et al. 2021), where the Nearctic and Neotropical bioregions meet. We selected other 'transitional' urban areas noted by MacGregor-Fors et al. (2021), occurring within our selected buffer area. We extracted information on study area, sampling design, and species lists from the studies that met our inclusion criteria.

We used reported species lists to generate a 'combined' dataset that also included the data for Zona Esmeralda. We kept details within their lists to generate 'data splits' to assess nuanced differences among studies. Spatial data splits included: references (negative or positive), residential or commercial areas, urban parks, ecotone side (natural or

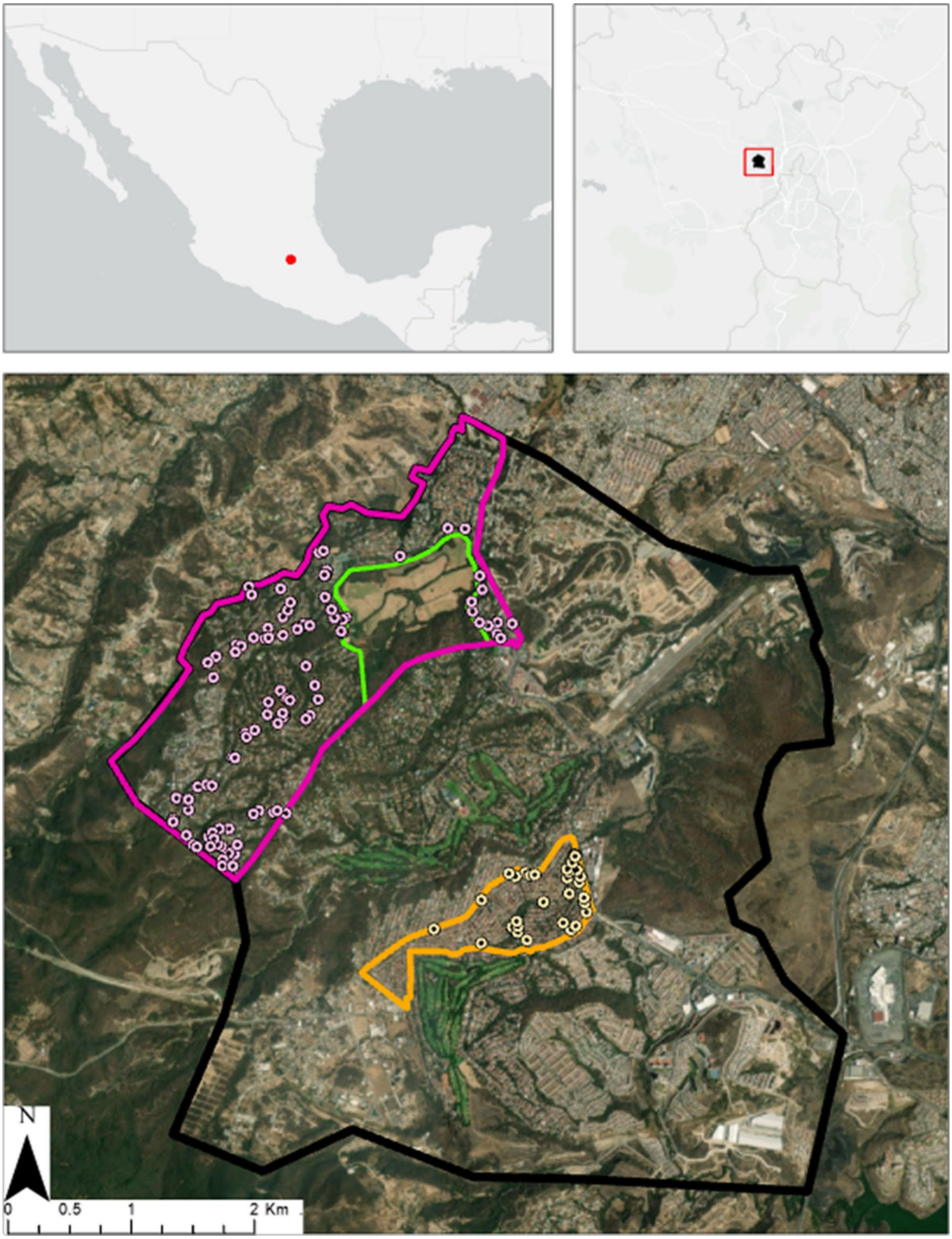

**Fig. 1** Study area. The approximate region is presented in red (upper maps) and Zona Esmeralda in black (all maps). The lower map shows Sayavedra in pink and Loma in yellow, with the polygons representing the extension of this housing developments and points representing the survey locations. The green polygon embedded within Sayavedra shows the boundary of its equestrian area. Background basemap and imagery obtained from ArcGIS Pro (ESRI 2021)

urban). Charre et al. (2013) used urban parks and 'reserves', we considered the latter as positive references. When available, temporal data splits included the season and community (breeding vs. nonbreeding). Knowing the relationship between studies, we attributed the temporal component to Buzo-Franco and Hernandez-Santin (2004) and used their spatial component to obtain detailed species lists attributed to González Oreja et al. (2007).

### Field surveys

We surveyed 'residential' areas, considering vacant lots and small public greenspaces within them (i.e. no-development areas and recreational parks). With a minimum area of 480 m$^2$, the survey locations varied in shape and size (but generally 12 × 50 m). Following González Oreja et al. (2007), we conducted 10-minute point count surveys with a radius of 25 m between 7am and 11 am. Each point location was visited once, and all surveys were conducted by one observer (LHS). Given the size difference between our housing developments, we used a stratified sampling approach. Survey locations were systematically photographed from the sidewalk and marked with a handheld GPS (eXplorist 500, Magellan®). We conducted one bird survey per location, using the fixed radius point count method first described by Hutto et al. (1986).

### Species identification

Species were identified by sight and song while in the field, using the 'Sibley guide to birds' (Sibley 2000), and reviewed upon gaining computer access. These reviews and name matching of the 'combined' dataset (for comparison among studies) were based on name conventions by the Cornell Lab of Ornithology (2024).

We modified one species record across the combined dataset. Studies reported one of three species of *Aphelocoma*: *A. ultramarina*, *A. californica*, or *A. coerulescens*. According to the Cornell Lab of Ornithology (2024) these three species share phenological similarities, but differ in distribution: the first is reported in central Mexico, the second in the California peninsula, and the third in Florida. Some or all of these individuals could instead be representatives of an additional species: *Aphelocoma woodhouseii*, a species that according to the Cornell Lab of Ornithology (2024) could also occur in the area. Nevertheless, we chose to change all species records to the previously recorded *Aphelocoma sp.* to avoid overrepresenting species diversity.

## Vegetation

### Vegetation variables

The survey location pictures were used to assess vegetation characteristics. For each location, we considered the 'Space Type' (no-development, vacant lot, or residential park), main 'Category' of the survey type (bare area, grass, mixed vegetation, or native vegetation), 'Strata' available (no cover, ground cover only, no midstorey, no overstorey, no ground cover, or three strata), and a developed a Complexity Score (CS). The CS was based on vegetation density and the type of vegetation available in each stratum. Complexity Score results ranged between 1 and 4, where the lowest score (CS1) was given to vacant lots with bare or limited ground cover and the highest complexity score (CS4) was given to remnant vegetation with multiple strata and high canopy cover (Fig. 2).

### Land classification

We classified land cover as either open vegetation (i.e. grass), closed vegetation (i.e. canopy cover), or impervious (e.g. housing and roads), using eCogniton (Trimble 2018). For this, we obtained satellite imagery from PlanetTeam (2017), which had a three meter pixel resolution, was cloud free in Zona Esmeralda, and reflected field conditions: acquisition date April 13, 2011. Then, we used ArcGIS Pro (ESRI 2021) to obtain sampling points every 50 m using the 'create fishnet' tool and assigned their corresponding land class using 'extract values' tool. For clarity, we call these as 'fishnet points'. The fishnet points excluded the 107 ha 'equestrian area' from Sayavedra to avoid its bias on the composition and availability of each land surface (see Fig. 1).

## Statistical analyses

Statistical analyses were conducted in RStudioTeam (2015).

## Avifauna

Considering only our field data, first we calculated spatial autocorrelation of our data using Moran's I using the package 'spdep' (Bivand et al. 2024). Moran's I results with values close to zero suggest non-significant spatial autocorrelation, while the sign of larger values suggest either positive (i.e. clustering) or negative autocorrelation (i.e. dispersion) (Bivand et al. 2009). Then, we used observed abundance data to estimate richness and generate species accumulation curves. Estimated richness was based on 'Jacknife 1', which reported as the most accurate non-parametric estimator for

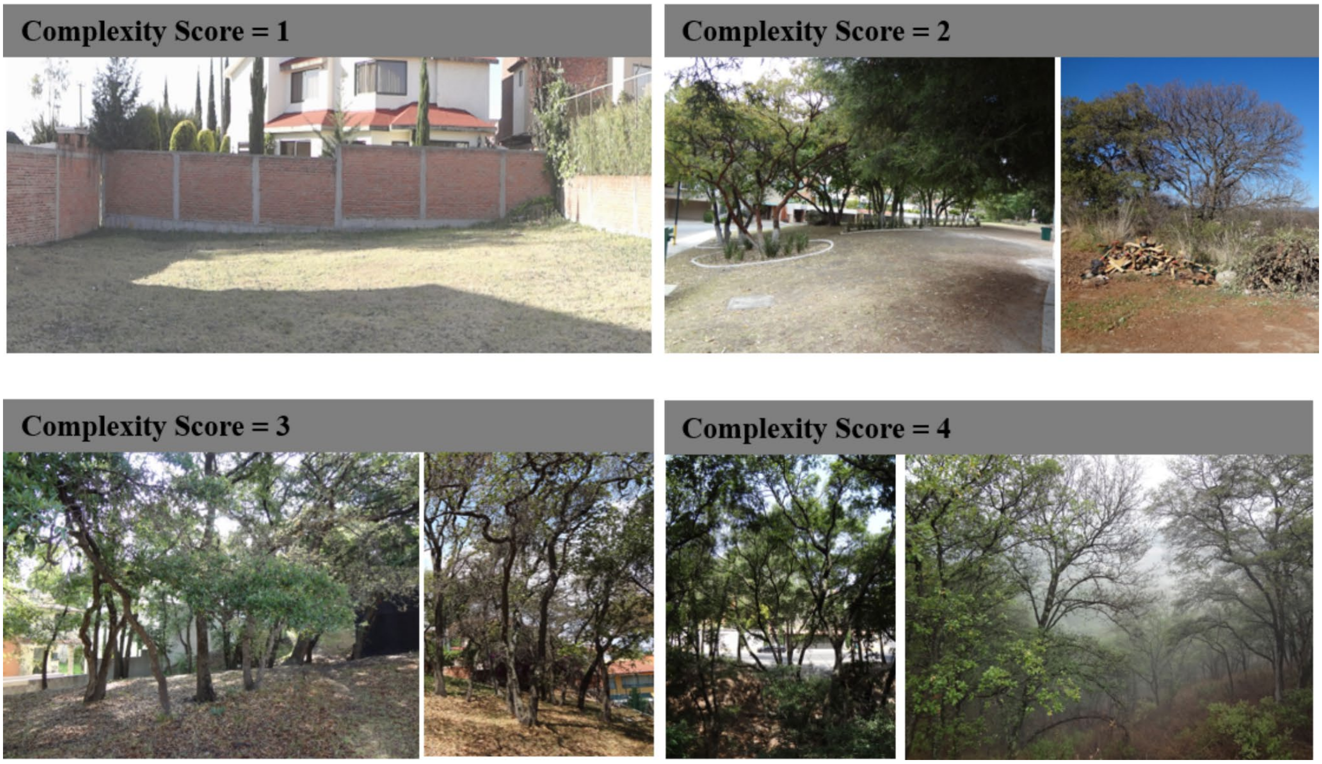

**Fig. 2** Complexity Score (CS) as attributed to different sites within the study. (1) CS1 was given to survey points where native vegetation has been removed leaving introduced grass or bare ground; (2) CS2 was given to survey points where more than 50% of the site had been cleared, with the remaining space containing native or introduced trees and/or shrubs; (3) CS3 was given to survey points with a more natural structure, represented by high incidence of trees and ground cover with leaves or grass; and, (4) CS4 was very similar to complexity 3 but presented a higher incidence of shrubs or young trees at the midstory level

studies similar to ours. Considering the comparison among studies, we used incidence data to calculate alpha diversity and beta diversity. Comparisons among studies were done for the overall data of a study and with any subdivision that we were able to identify (e.g. seasonal, urban parks, negative reference, residential, etc.). We used Shannon's entropy index as a measure of alpha diversity and Sorensen's index of co-occurrence for beta diversity. Higher values for Shannon's index suggest higher diversity, while those for Sorensen's indicate greater similarity. Analyses were conducted using the package 'BiodiversityR' (Kindt 2024).

## Habitat characterization

We used a $\chi^2$ to understand if vegetation complexity or land cover surfaces occurred as would be expected by chance within our sites. We used the signs of the residuals to interpret results, indicating a greater (positive sign) or lower (negative sign) frequency observation than would be expected. Vegetation complexity used the CS score given to each survey point, while that for land cover surfaces used the fishnet points.

## Relationship between avifauna and vegetation

We generated a series of Poisson-based general linear models (GLMs) to test if species richness (response variable) could be explained by the different vegetation variables available or site (predictor variables). These GLMs tested all possible combinations of variables. We used the Akaike's Information Criterion corrected for small samples ($AIC_c$) to determine the top five models, which we present in results. Lower $AIC_c$ values suggest stronger relationships between response and predictor variables, relative to other models. To assess model performance, we calculated the difference in $AIC_c$ values between the first models and the rest ($\Delta AIC_c$), with $\Delta AIC_c < 2.0$ representing models with substantial empirical support (Burnham and Anderson 2003). We discarded models with $\Delta AIC_c > 2.0$, which have lower support.

**Table 1** Summary of results obtained during the literature review. In method, r = radius

| | | Zona Esmeralda | Buzo-Franco and Hernandez-Santin (2004) | Charre et al. (2013) | González-Oreja et al. (2007) | Ortega-Alvarez & MacGregor-Fors (2009) | Puga-Caballero & McGregor-Fors (2014) |
|---|---|---|---|---|---|---|---|
| Study area | City | Mexico City | Puebla City | Mexico City | Puebla City | Mexico City | Mexico City |
| | within city | northwest | throughout | throughout | throughout | southwest | northeast |
| | Space type | residential -and parks and natural areas inside | urban parks and negative reference | urban parks & reserves | urban parks and negative and positive reference | commercial, residential, green areas | peri urban ecotones, and negative and positive reference |
| | Area range (ha) | 91–478 | 0.66 to 109 | 11-1100 | NA | NA | NA |
| Study design | Method | 10 min; points *r* = 25 m | 10 min; point *r* = 25 m | 10 min; points *r* = 20 m | 10 min; point *r* = 25 m | 20 min; point r = unlimited | 5 min; point r = unlimited; points along transects |
| | No. months | 3 | 8 | 8 | 1 | 3? | 1 |
| | Seasons | spring, summer | all | winter, spring, summer | spring | summer | summer |
| | No. surveys | 148 | 1,378 | 125 | 355 | 160 | 180 |
| Results | Spp. richness | 56 | 62 | 96 | 51 | 58 | 43 |

## Results

### Literature review

We found five studies that met our inclusion criteria, with one of them representing a thesis (Table 1). The thesis by Buzo-Franco and Hernandez-Santin (2004) presented a temporal component (seven sites visited one per month over eight months) and a spatial component (21 sites visited once over one month; including the seven sites of the temporal component). The spatial component was published by González Oreja et al. (2007). Thus, to avoid data duplication, we assigned only the temporal chapter to Buzo-Franco and Hernandez-Santin (2004) and the spatial component to González Oreja et al. (2007).

### Avifauna

Over the course of three months (2-Apr-2012 to 30-Jun-2012), we conducted a total of 148 point-surveys in Zona Esmeralda, where we identified 1,007 individuals from 56 species (44 resident and 12 migrant species). In addition, we were able to partially identify 71 individuals that were assigned to 13 partially identified species using annotations made while in the field. This represents 10 partially identified species in Sayavedra and 5 in Loma. We excluded partially identified species from the analysis.

Of the 56 species identified, we found 52 species (766 individuals) in Sayavedra and 26 species (241 individuals) in Loma. Based on the 'Jacknife 1' estimator, species richness was estimated at 67 species for Sayavedra and 33 for Loma, with an overall estimated richness of 71 species for Zona Esmeralda. We found no evidence of spatial autocorrelation considering data based on abundance (Moran's I = 0.013, *p* = 0.352) or species richness (Moran's I = 0.036, *p* = 0.215). These results suggest that neither abundance or species richness had spatial clustering or dispersion, with patterns that were not significantly different from what would be expected by chance. Despite the difference between observed and estimated richness, the species accumulation curves showed that the sampling effort was adequate (Fig. 3). The gap between the observed and estimated accumulation curves suggest that we had a successful detection rate of 79% of the birds occurring in Loma and 78% of those in Sayavedra, suggesting similar sampling effort between our sites and adequate comparability within our study area.

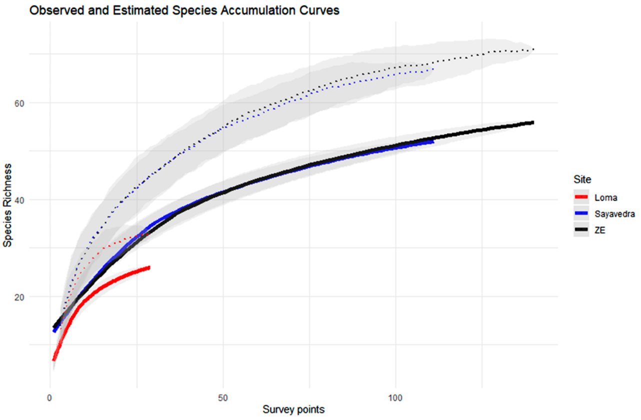

**Fig. 3** Species accumulation curve for Zona Esmeralda considering observed (bold lines) or estimated (dotted lines) avifauna richness. Grey areas represent the 95% confidence intervals. Red lines represent Loma, blue lines represent Sayavedra, and black lines represent Zona Esmeralda

**Table 2** Results for alpha and beta diversity indices among studies reviewed. These calculations were based on presence-only data, using available species lists retrieved during the literature review and split into subdivisions differentiated by the respective authors. Beta diversity is only presented in relation to Zona Esmeralda and its site (Sayavedra and Loma). Shannon's entropy is considered a measure of alpha diversity; Sorensen's = beta diversity; RefNeg = negative reference; RefPos = positive reference; UrbanSide = urban side of the ecotone; NatSide = natural side of the ecotone

| Data source | Data split | Shannon's index | Sorensen's index | | |
|---|---|---|---|---|---|
| | | | Zona Esmeralda | Sayavedra | Loma |
| Buzo-Franco and Hernandez-Santin (2004) | Overall | 4.13 | 0.44 | 0.42 | 0.41 |
| | RefNeg | 2.48 | 0.29 | 0.25 | 0.42 |
| | Urban parks | 4.13 | 0.44 | 0.42 | 0.41 |
| | Winter | 3.76 | 0.48 | 0.46 | 0.49 |
| | Spring | 4.09 | 0.45 | 0.43 | 0.42 |
| | Summer | 3.87 | 0.48 | 0.46 | 0.46 |
| | Fall | 3.74 | 0.47 | 0.45 | 0.47 |
| | Breeding | 4.11 | 0.44 | 0.42 | 0.41 |
| | Nonbreeding | 3.89 | 0.48 | 0.46 | 0.48 |
| Charre et al. (2013) | Overall | 4.56 | 0.47 | 0.43 | 0.34 |
| | RefPos | 4.37 | 0.5 | 0.46 | 0.38 |
| | Urban parks | 4.38 | 0.47 | 0.44 | 0.38 |
| González Oreja et al. (2007) | Overall | 3.95 | 0.5 | 0.48 | 0.49 |
| | RefPos | 3.58 | 0.5 | 0.48 | 0.52 |
| | RefNeg | 1.79 | 0.16 | 0.14 | 0.25 |
| | Urban parks | 3.91 | 0.47 | 0.45 | 0.47 |
| Ortega-Álvarez and MacGregor-Fors (2009) | Overall | 4.06 | 0.49 | 0.47 | 0.43 |
| | Urban parks | 3.97 | 0.46 | 0.46 | 0.41 |
| | Residential | 3.5 | 0.47 | 0.47 | 0.51 |
| | Commercial | 3.18 | 0.5 | 0.5 | 0.48 |
| Puga-Caballero et al. (2014) | Overall | 3.76 | 0.48 | 0.44 | 0.38 |
| | RefNeg | 1.95 | 0.22 | 0.2 | 0.24 |
| | RefPos | 3.04 | 0.42 | 0.41 | 0.38 |
| | UrbanSide | 3 | 0.39 | 0.36 | 0.43 |
| | NatSide | 3.58 | 0.43 | 0.41 | 0.35 |
| Zona Esmeralda | Overall | 4.03 | | 0.96 | 0.63 |
| Sayavedra | Residential | 3.95 | | | 0.56 |
| Loma | Residential | 3.26 | | | |

## Alpha and beta diversities among studies

Results for alpha and beta diversities are presented in Table 2. We found that Sayavedra had greater diversity than Loma, with an overall Shannon's entropy index value of 3.01 for Zona Esmeralda. Sayavedra and Loma shared over half of their diversity (Sorensen's index = 0.56).

When comparing studies, we found that Charre et al. (2013) had the greatest alpha diversity (from Shannon's index = 4.56) and Puga-Caballero et al. (2014) had the lowest (3.76). Considering the overall data for each study, Zona Esmeralda shared the most species with González Oreja et al. (2007) and the least species with Buzo-Franco and Hernandez-Santin (2004) (Sorensen's index = 0.5 and 0.44, respectively). Splits by data type revealed that Loma shared the most species with the positive reference by (González Oreja et al. 2007) and Sayavedra shared the least species with the negative reference of the same study (Sorensen's index 0.52 and 0.14, respectively). Comparisons across all studies and their data splits can be found in Supplementary Material.

**Table 3** Vegetation attributes observed in Zona Esmeralda, separated by site. Based on fieldwork, vegetation complexity scores (CS) represent the number of point-count surveys per vegetation category. Lower scores represent simpler vegetation assemblages, while higher scores represent complex ones that tend towards natural assemblages. Based on remote sensing, vegetation availability represents the number of fishnet points per surface cover class, per housing development

| | Vegetation category | Loma | Sayavedra |
|---|---|---|---|
| Vegetation complexity | CS1 | 15 | 2 |
| | CS2 | 9 | 13 |
| | CS3 | 11 | 32 |
| | CS4 | 1 | 65 |
| Vegetation availability | impervious | 171 | 280 |
| | open vegetation | 18 | 169 |
| | closed vegetation | 177 | 1448 |

## Habitat characterisation

*Vegetation complexity.* The number of point surveys per complexity score per site suggest that Sayavedra had greater vegetation complexity, while Loma had a tendency for lower vegetation complexity scores (Table 3). These differences were statistically significant ($\chi^2_{df=2} = 59.70, p < 0.001$). Closer inspection of the residuals showed that the number of point surveys on the complexity categories CS1, CS2, and

**Table 4** Poisson-based general linear models created to test whether the number species (response variable) was explained by a different array of vegetation characteristics and site (predictor variables). $AIC_c$ = Akike's information criterion corrected for small samples, se = standard error; df = degrees of freedom. *Denotes factors where at least one level was significant

| Model No. | Predictor variables | $AIC_c$ | $\Delta AIC_c$ | intercept | | | | Residual deviance | DF |
|---|---|---|---|---|---|---|---|---|---|
| | | | | Estimate | SE | Z value | *p* value | | |
| 1 | Space Type | 595.74 | - | 1.15 | 0.09 | 12.16 | <0.001 | 186.62 | 145 |
| 2 | Complexity*, Space Type | 596.45 | 0.71 | 0.78 | 0.21 | 3.78 | <0.001 | 180.91 | 142 |
| 3 | Site, Space Type | 597.24 | 1.50 | 1.08 | 0.14 | 7.72 | <0.001 | 186.01 | 144 |
| 4 | Space Type*, Category | 597.40 | 1.66 | 1.02 | 0.35 | 2.89 | <0.01 | 181.85 | 142 |
| 5 | Site, Complexity*, Space Type* | 598.32 | 2.58 | 0.78 | 0.21 | 3.77 | <0.001 | 180.57 | 141 |

CS3 was higher than expected for Loma (Residuals = 5.34, 1.58, 0.17, respectively) and lower than expected for Sayavedra (-3.03, -0.89, and −0.09, respectively), while that of CS4 was lower than expected in Loma (Residuals = -3.74) and higher than expected in Sayavedra (2.13). The difference was most marked differences occurred for CS1 (representing 42% of the points in Loma and 2% of the points in Sayavedra) and CS4 (3% of the points in Loma and 58% of the points in Sayavedra), and least marked for CS3 (31% in Loma and 29% in Sayavedra).

*Vegetation availability.* Despite both sites having greater availability of vegetated surfaces, we found significant differences in the availability of each land class across sites ($\chi^2_{df=2}$ = 196.60, $p < 0.001$). In fact, the comparison of fishnet points showed that Loma had only marginally greater availability of vegetated surfaces (1.1 times more vegetation than impervious), while Sayavedra had 5.8 greater availability of vegetated surfaces than impervious ones (Table 3). Accordingly, the adjusted residuals indicated that Loma had lower availability of closed and open vegetation (residuals = -5.39 and −2.23, respectively) and much higher availability of impervious surfaces (residuals = 11.48) than expected. Conversely, Sayavedra had a negative association with impervious surfaces (residuals = -5.04) and a positive one with closed and open vegetation (residuals = 2.33 and 0.98, respectively).

### Relationship between avifauna and vegetation

Table 4 shows the top five models, which had a positive relationship between some of the vegetation characteristics measured and species richness. We found good model fit among the first four models ($\Delta AIC_c < 2.0$); in contrast, the fifth model may not be as strongly supported as the others ($\Delta AIC_c = 2.58$; and therefore not discussed further).

Considering the levels within the predictor variables among the models with good fit, we found only two variables that yielded significant differences: Complexity in Model 2 and Space Type in Model 4. Considering that Space Type occurred without significant differences in Models 1 to 3, its effect may be context-dependent or less consistent across different model specifications compared to Complexity. For Complexity (Model 2), we found a positive effect of higher Complexity Scores (CS3: Estimate = 0.382, $p = 0.039$; and CS4: Estimate = 0.371, $p = 0.049$) on species richness, indicating that areas with more complex vegetation structure support greater species diversity.

## Discussion

To evaluate the benefits from biodiversity inclusive design, even when incidentally used, we focused on one of the most important steps for this design practice: assessing species composition and the factors that influence it (Hernandez-Santin et al. 2022). We used the ecological context to understand which observations could be attributed to known drivers within urban ecology. Ultimately explaining the differences between our sites, we found that higher avifauna diversity in Sayavedra was supported by greater vegetation complexity, higher canopy cover, and lower impervious surface cover than Loma. Such differences in the availability of land classes within Zona Esmeralda can be attributed to the historical built-design regulations and prevalent upkeep (maintenance) of vegetated areas. Then, we discuss the implications of our ecological observations in context of biodiversity inclusive design.

### Ecological context

We used a meta-analysis to understand the diversity of avifauna in Zona Esmeralda in context with other studies that have been conducted in the region. We assessed aspects relating to the temporal component (seasonality) and the spatial component (fragmentation, heterogeneity, and urbanisation gradients). Findings from the ecological component of our study suggest that the value of greenspaces across the urbanisation gradient can be attributed to its habitat characteristics including vegetation complexity and the availability of impervious surfaces and canopy cover. This agrees with previous studies, such as the metanalyses by Nielsen et al. (2014) and Beninde et al. (2015), where habitat heterogeneity have been identified as the most important driver of biodiversity in urban areas. We acknowledge that urbanisation history (i.e. age since urbanisation) could influence biodiversity patterns over time. However, data limitations prevented us from incorporating it directly into our models.

### Seasonality

Our observations from the seasonality component suggest that the diversity of Zona Esmeralda should be compared to data considering the breeding season alone, or cautiously compared to studies considering both seasons. Based on studies that considered both seasons, we found support for the hypothesis that studies monitoring the breeding (spring and summer) and nonbreeding (autumn and winter) communities would have greater alpha diversity, but had mixed support that the breeding community was more diverse. The studies by Charre et al. (2013) and Buzo-Franco and Hernandez-Santin (2004) who monitored avifauna for 8 months had the highest species diversity (Shannon's entropy: 4.56 and 4.13, respectively) and also one of the highest beta diversity values (Sorensen's = 0.52) despite being conducted in different cities. The patterns of diversity per season could be explained by scale. At the broader scale (study), we found the expected higher diversity during the breeding season. However, at the site scale erratic patterns were observed. For example, of the 12 sites monitored by Charre et al. (2013), the authors reported seven sites with higher richness in winter (including one reserve), four in summer (including three reserves), and one did not change (the smallest urban park: 11 ha).

Focusing only on the breeding season across studies, we found that seasonality alone did not explain differences in alpha diversity. For example, while we found that Buzo-Franco and Hernandez-Santin (2004) had greater alpha diversity in spring than summer, but Ortega-Álvarez and MacGregor-Fors (2009) had greater diversity in summer than that of Zona Esmeralda, González Oreja et al. (2007), or Puga-Caballero et al. (2014), in descending order. Nevertheless, we found that, while the monitoring period within the breeding community may not drive species richness, it may influence species composition to some degree. For example, our study area shared the most species with the study by González Oreja et al. (2007), which is the only other study that explicitly monitored spring (Sorensen's = 0.50).

### Fragmentation gradient

We found support for the hypothesis that studies with greater spatial proximity would be expected to share more species. Zona Esmeralda, located in the northwest of Mexico City shared the most species with studies focusing on the same season and city, including Ortega-Álvarez and MacGregor-Fors (2009) located in the southwest of the city and Puga-Caballero et al. (2014) in the northeast (Sorensen's = 0.49 and 0.48, respectively).

Considering the connectivity gradient, Charre et al. (2013) reported that species richness was better explained by park size (with larger parks holding higher richness) than connectivity. Nevertheless, the metanalysis by Beninde et al. (2015) concluded that enhancing connectivity within urban areas (e.g. through corridors) was one of the most important aspects to support high levels of species richness. Thus, high availability of canopy cover in Zona Esmeralda (based on remote sensing analysis) and relatively complex patches (based on vegetation complexity analysis) within housing developments could help to dilute the effects from fragmentation and contribute to enhance widespread connectivity.

### Heterogeneity and urbanisation gradients

We found that the ecological value of Zona Esmeralda was higher than that traditionally placed on residential areas, with differences between our housing developments. Residential areas are often considered to have lower ecological value than other greenspaces. In fact, rather than greenspaces, MacGregor-Fors et al. (2021) considered residential areas as 'heavily-built' areas. Similarly, White et al. (2005) found greater habitat heterogeneity and avifauna diversity in parks than in streetscapes with native features, or in highly modified streetscapes with non-native components (most simple habitats with least avifauna). Among the studies we reviewed, Ortega-Álvarez and MacGregor-Fors (2009) found that residential areas represented the middle point between greenspaces and commercial areas in terms of both avifauna composition and habitat characterisation. When considering data splits from studies focusing on the breeding community, we found that Zona Esmeralda and sometimes Sayavedra outperformed greenspaces (urban parks and positive references); while Loma only outperformed negative references with one exception. The exception when Loma was compared to Puga-Caballero et al. (2014), which had lower values for the urban portions of their ecotones and their positive and negative references.

Since we found that the portion of the urbanisation gradient might be driving differences in diversity observed during this study, we reviewed habitat characteristics. We suggest that surface cover of foliage and impervious classifications explained differences in diversity. This is supported by our statistical analyses, where we found that the availability of impervious cover and of canopy cover was statistically different between our sites. In addition to the percentage of surface cover, the ecological value of greenspaces may be attributed to vegetation complexity. In fact, it has been noted that, while all greenspaces in urban areas have the potential to support biodiversity, the contribution of greenspaces depends on the spatial features that affect their habitat quality (Beninde et al. 2015).This is also supported by our general linear model with species richness significantly driven by vegetation complexity of the survey points, where areas with greater vegetation complexity had greater vegetation complexity scores.

## Biodiversity inclusive design context

### Regulations

Each residential area created its own process and regulations for home development within their boundaries (Capron and Esquivel Hernández 2016). When Sayavedra and Loma were first established, construction licenses were provided by the local government but regulated by each housing development. Other than the legal requirement for ‘no-development’ assigned to riparian areas within each housing development, the municipality granted licences with the caveat that the design should be pre-approved by the residential areas where each house would be built (Jimenez Cantu 1979). This gave housing developments the control over the character of homes constructed within their boundaries.

Sayavedra and Loma included regulations that purposely or incidentally mitigated the effects of urbanisation from social and ecological perspectives. For example, they both dictated that each property should hold a detached house with gabled roofs (with 20–25% slopes) (Condado de Sayavedra; Loma de Valle Escondido). From the social perspective, these regulations allowed aesthetic cohesion within the gated community, with space around houses further ensuring the reduction of noise, and visual crossover between properties. From the ecological perspective, this retained site permeability and allowed for larger areas with vegetation and its associated wildlife, giving space to biodiversity and supporting landscape connectivity (Beninde et al. 2015).

Nevertheless, Sayavedra was undoubtedly more biodiversity friendly. For example, while they both dictated the need for drainage specific to rainwater (Condado de Sayavedra; Loma de Valle Escondido), obligations in Sayavedra ensured diversion into the riparian systems naturally occurring within the development or natural infiltration that allowed replenishing groundwater systems, such as specifying the requirement of 40% pervious surfaces and the use of French drains along sidewalks. Therefore, Sayavedra’s features further mitigated disturbances to the natural water cycle and promoted the use of vegetated yards (back and front). Similarly, both sites had requirements for specific construction materials (Condado de Sayavedra; Loma de Valle Escondido). However, Sayavedra further specified that materials should be locally sourced, that vegetation should be protected during construction phases, and that vegetation should be upkept using biodegradable pesticides and fertilisers.

### Vegetation upkeep

The housing developments also differed in their obligations regarding native trees or the upkeep of vegetated areas, leading to differences in the vegetation composition and structure at each site. Both sites had an economic penalisation for every tree removed during the construction phase -or any time after that- (Condado de Sayavedra; Loma de Valle Escondido). This could suggest an early (or unintended) attempt to monetarise ecosystem services provided by trees and to have an incentive to minimise native vegetation removal. Regardless, the effect from such penalisations had inherent differences in their success as mitigation practices, due to the starting point of vacant lots (i.e. pre-construction). Sayavedra partitioned into lots that kept their original (native) vegetation assemblages, with vegetation upkeep mostly restricted to public areas such as median strips and small parks for recreational use. This means that property owners looking to build had ample native trees to design around or pay hefty tree-falling penalties, resulting in a neighbourhood with many wildlife-friendly gardens (personal observation). In addition, Sayavedra limited human access to ‘no-development’ areas, by fencing them off. This activity inherently benefitted biodiversity. In contrast, Loma pre-cleared most of the native vegetation from its subdivision lots and revegetated using exotic grasses that continue to be regularly mowed. Thus, vegetation upkeep in Loma was shared across vacant lots and public areas (including median strips and small parks for recreational use), offering a ‘tabula rasa’ (i.e. ‘blank slate’) to prospective owners and with limited availability of ‘no-development’ areas that provided access to pedestrians. This resulted in more houses with traditionally manicured gardens (personal observation). Thus, while it could be argued whether observed vegetation assemblages respond to economical penalisations or aesthetic preferences shared among people living there, Sayavedra undoubtedly bestowed the prevalence of vegetation attributes.

Based on the regulatory and vegetation upkeep strategies, Sayavedra would be expected to have greater vegetation attributes. This was supported by our results, with significant differences in vegetation complexity and availability between our sites. Vegetation complexity scores were simpler in Loma (e.g. CS1, the simplest score, represented 42% of the survey points in Loma and 2% in Sayavedra) and more complex in Sayavedra (e.g. CS4, the most complex score, represented 3% in Loma and 58% in Sayavedra). Similarly, Sayavedra had a canopy cover of 76% and impervious surfaces over 15% of its extension (excluding the equestrian area), while Loma had a canopy cover of 48% and impervious surfaces over 47% of its extension. Notably, these percentages of land surfaces exclude impervious surfaces 'hidden' under the canopy of trees. This suggests that the quasi-continuous canopy cover may be an important source of connectivity from the avifauna perspective. More importantly, these differences in vegetation-related strategies ultimately contributed to the outperformance of avifauna diversity by Sayavedra over Loma, and may similarly contributed to its outperformance over multiple studies reviewed (particularly when considering the data splits: Table 2).

Construction regulations and vegetation upkeeping strategies contributed to enhance avifauna diversity of Zona Esmeralda resulting in greater diversity than would be expected from similar areas, as shown through our assessment of the ecological context. These benefits occurred without purposely using practices expected from frameworks aligned with biodiversity inclusive thinking. For example, under themes of biodiversity inclusive design, there would have been a need to assess biodiversity before the development of these gated communities, establish clear biodiversity targets (or ecosystem services) to be protected, or ensure resource availability to fulfil wildlife requirements in their areas (Apfelbeck et al. 2020; Garrard et al. 2018; Hernandez-Santin et al. 2022; Weisser and Hauck 2017).

Our study demonstrates that even incidental use of biodiversity inclusive design principles can contribute to enhance avifauna diversity, with differences between our sites highlighting the fragile balance of species prevalence. Our results showed grater biodiversity benefits within Sayavedra, which had stricter regulations to accommodate natural areas. This is true from Sayavedra's pre-construction and construction phases to their current vegetation upkeeping strategies. In contrast, data splits showed that Loma shared similarities with sites subject to greater anthropogenic pressures. Thus, we emphasise the importance of minimising disturbances to natural vegetation in vacant lots and during construction. As shown, areas would be most benefited by prioritising the requirement of naturally vegetated areas as starting point for constructions. Nevertheless, the actions used fell short of providing the full potential of biodiversity inclusive thinking, which have the potential to substantially increase benefits for both biodiversity and people when purposely applied through activities such as those outlined by Bekessy et al. (2020) at the property scale.

Our results also suggest that the intentional use of biodiversity inclusive principles over larger scales such as ours (or larger) would be most beneficial within urban areas. Our assessment involved the housing development scale, where we found evidence that the widespread use of a limited number of biodiversity inclusive principles contributed to enhance the ecological value of a whole residential area (using bird diversity as a proxy of such value), even when inadvertently used. However, the application of these principles at the property scale may not allow to counteract landscape scale effects from urbanisation. Thus, we urge the large-scale incorporation of biodiversity inclusive thinking through regulatory measures by private entities, such as those described here, or by governmental agencies. Thus, our study supports the global calls for neighbourhood and city-scale biodiversity inclusive planning (Oke et al. 2021). Although conclusive evidence of the benefits by purposely using biodiversity inclusive thinking is still pending, modelling outcomes have contributed to show the potential of these frameworks. For example, an urban renewable project in Fishermans Bend (located in the core of Melbourne, Australia) used connectivity models for a variety of species from fauna groups with different mobility capabilities, and found that the use of biodiversity inclusive principles at the neighbourhood scale offered promising outcomes for species connectivity while simultaneously boosting liveability, health, and wellbeing for people (Kirk et al. 2021).

Although our housing developments were developed ~ 40 years ago, relatively recent changes in regulations make the next 10 years crucial to ensure the persistence of species found during our study. Our study highlights the fragile balance of avifauna diversity within Zona Esmeralda, explained by differences in vegetation complexity, vegetation availability, and the percentage of impervious surfaces. However, relatively recent changes (2003) in construction regulations at the State level are preventing residential areas from enforcing their biodiversity-friendly regulations (Gaceta del Gobierno 2003). For example, new regulations allow offsetting tree removals by plating 24 juvenile trees, with at least 12 of those within the housing development (Roberto Aviles, 2019, personal communication). This regulation change is unfortunate because juvenile trees cannot provide the same ecosystem functions as mature trees, with high mortality rates in urban areas expected for trees in age classes of three to seven years, and surviving trees taking decades to fully develop their functions (Brunner and Cozens 2013; Lindenmayer et al. 2000; Standards Reference Group SERA 2017). Thus, 'recent' changes in construction regulations may heighten the effects of urbanisation across Zona Esmeralda and may contribute to break the patterns of avifauna diversity observed in our study. This is alarming for Sayavedra, which seemed to have mostly escaped the negative effects from urbanisation, maintaining greater species diversity than is traditionally assigned to residential areas. Meanwhile, for Loma, 'recent' regulatory changes could lead to a steep decline in species richness, along with an increase in the occurrence, spatial distribution, and abundance of urban exploiter species. Therefore, we urge to revert legislative and management regulations in Zona Esmeralda to maintain the biodiversity observed during our study. Actively incorporating practices of biodiversity inclusive thinking could help maintain this relatively well-preserved environment that may act as refuge for wildlife in urban environments.

## Conclusion

The literature review and comparative diversity assessments allowed to provide ecological context for the observations in our study area (Zona Esmeralda) and the potential differences between our sites (Sayavedra and Loma). Within the ecological context, our results showed that the main drivers of differences in bird diversity of the breeding community were related to heterogeneity and urbanisation gradients, more than to sampling design, seasonality (when considering the breeding season only), or fragmentation. Notably, seasonality in relation to sampling design (i.e. monitoring period) was an important predictor of differences between studies monitoring the breeding community and those monitoring the breeding and nonbreeding communities.

Within the biodiversity inclusive thinking context, we found that differences between our sites were best explained by the regulations and vegetation upkeeping strategies used to maintain vegetated areas in general, but particularly vacant lots. Our study showed an example of incidental use of principles aligned with practices relating to biodiversity inclusive design (or thinking, more broadly), where vegetation upkeep strategies (during the construction and maintenance periods) were the strongest contributor to a diverse avian community. Our study highlights that even applying a few principles of biodiversity inclusive design, particularly dealing with vegetation upkeep in urban landscapes, has the potential to benefit wildlife by fostering greater diversity of species than would be expected from similar spaces that do not apply such principles. Thus, biodiversity inclusive thinking has the potential to increase the ecological value of properties at all scales, with greater importance of such ecological boost when applied over larger scales such as the housing development one as observed during our study. This is particularly true for avifauna, our focus group, but could potentially lead to similar results in other fauna groups relative to their expected response to urbanisation. For example, although ground-dwelling species would be expected to have higher impacts form impervious surfaces even within Sayavedra, the effects from these features may still be less prevalent in Sayavedra than in other residential areas without incidental or explicit use of principles aligned with biodiversity inclusive thinking.

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