Universidad Autónoma de Querétaro

Facultad de Ciencias Naturales

Diversidad de aves en áreas verdes urbanas de México: una aproximación multiescala

Tesis

Que como parte de los requisitos
para obtener el Grado de

Doctor en Ciencias Biológicas

Presenta

Remedios Nava Díaz

Dirigido por:

Dr. Rubén Pineda López

Querétaro, Qro., a 28 de abril de 2021

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Dr. Rubén Pineda López
Presidente

Dr. Luis Gerardo Hernández Sandoval
Secretario

Dra. Iriana Leticia Zuria Jordan
Vocal

Dr. Israel Gustavo Carrillo Angeles
Suplente

Dr. Robert Wallace Jones
Suplente

Centro Universitario, Querétaro, Qro.
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DEDICATORIA

A mi papá. Gracias por todo.

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RESUMEN

La urbanización es una de las múltiples amenazas de la biodiversidad. El impacto negativo de la urbanización sobre la biodiversidad se debe a diversos mecanismos entre los que destaca la modificación del hábitat. Los cambios en el hábitat suelen ser intensos e irreversibles lo que provoca que numerosas especies reduzcan sus números o se extingan localmente. Sin embargo, las zonas urbanas pueden albergar una fracción importante de las especies que ocupaban el área antes de que esta fuera urbanizada e incluso pueden favorecer a un número reducido de especies. Dentro de las zonas urbanas, las áreas verdes pueden mantener niveles de diversidad altos. Por tal motivo, se ha realizado un gran esfuerzo académico por comprender los mecanismos que modulan la diversidad de las áreas verdes urbanas. El objetivo del presente trabajo fue evaluar la influencia de variables ambientales que describen la modificación del hábitat sobre la diversidad de aves en áreas verdes a diferentes escalas espaciales. Si bien este tema ya ha sido investigado anteriormente, la aproximación de este trabajo es distinta. Se emplearon métodos de análisis poco recurridos en los estudios de ecología urbana que incluyen el uso de información sobre los rasgos funcionales de las especies, así como medidas de diversidad adicionales a la taxonómica y por medio de la aleatorización de éstas, se evaluó si la estructura funcional y filogenética de los ensambles de aves. Se decidió emplear al grupo de las aves por ser un grupo que cuenta con suficiente información para contrastar los resultados obtenidos. En el primer capítulo de este trabajo se reunieron las publicaciones sobre el estudio de la avifauna en áreas verdes de México. El fin de la revisión fue conocer los temas explorados en la materia hasta entonces para proponer líneas de investigación que complementaran el conocimiento existente. Como resultado de la revisión de literatura, se plantearon las siguientes preguntas que fueron exploradas en el segundo capítulo: ¿la respuesta de las especies a la vegetación nativa y a la vegetación exótica es similar? y ¿cuál es el papel de la matriz que rodea a las áreas verdes? En el tercer capítulo se estudió detalladamente la respuesta de la diversidad de aves a características de las áreas verdes definidas a dos escalas: el punto de conteo y el parche. En este último nivel espacial, se incluyó información sobre la matriz que rodea el parche. En este capítulo se usaron tres medidas complementarias de diversidad con el fin de tener un panorama más amplio del

fenómeno. Como parte del capítulo se probó si la transformación del hábitat en las áreas verdes actuaba como un filtro ambiental.

Los resultados de este trabajo señalan que las especies vegetales nativas son relevantes para las aves especialistas y que las especies residentes se distribuyen en las áreas verdes en función de sus rasgos y las características de los sitios, a diferencia de las especies migratorias que no mostraron un patrón significativo. Los resultados obtenidos en este trabajo confirman que la matriz juega un papel ecológico fundamental y señalan que su efecto puede ser mayor que otras características ambientales al interior del área verde. Por otro lado, si bien se registró una reducción en las diferentes medidas de diversidad en los sitios más transformados, las pruebas empleadas no confirman un proceso de filtraje ambiental en las áreas verdes estudiadas. De igual manera, se mostró que las relaciones entre la diversidad de aves y las características ambientales de las áreas verdes, evaluadas a nivel de punto de conteo y de parche, varían temporalmente.

ABSTRACT

Urbanization stands as one of the multiple threats to biodiversity. Urbanization negative impact on biodiversity is due to several mechanisms among which habitat modification stands out. Habitat changes tend to be intense and irreversible, and this leads to reductions in species populations or to their local extinction. However, urban areas can harbor a large fraction of species present in the area before it was urbanized and they can even favor a reduced number of species. Within urban areas, green spaces can maintain high diversity values. Given this fact, a large academic effort has been done to understand the mechanisms that drive diversity in urban green spaces. This study is aimed to assess the influence of environmental variables that describe habitat modification on bird diversity within urban green spaces considering different spatial scales. Though this issue has been previously investigated, this work approach is different. Scarcely used methods in urban ecology that rely on species trait information were employed, and different measures of diversity, complementary to taxonomic diversity, were used and through randomizations of these, the functional and phylogenetic structure of bird assemblages was assessed. Birds were selected as the study group because they have been thoroughly studied and hence, enough information is available to compare herein generated results. In chapter I, the publications about the study of urban birds in green spaces of Mexico were gathered. This review was intended to give an insight into already explored issues so further research questions could be made. As a result, the following questions were addressed in chapter 2: do birds respond in a similar way to native plant species and exotic plant species? and which is the role of the matrix surrounding green spaces? In chapter 3 the answer of bird diversity to green spaces environmental characteristics was studied considering two scales: the count point and the patch. The patch level included information about the matrix around studied green spaces. In this chapter, three diversity measures were employed in order to get a deeper understanding of the phenomenon. This chapter also included an assessment of the environmental filtering due to habitat transformation within green spaces.

This work results show a relevant role of native vegetative for specialist bird species and that the distribution of resident birds depends on their functional traits and the green spaces characteristics, while migratory species distribution does not. Results herein reported confirm the matrix significant role and show that its effect could be even larger than the effect of other green spaces characteristics. Moreover, despite the fact a reduction of diversity in the most transformed sites was registered, the statistical tests did not support the environmental filtering of bird assemblages in the studied green spaces. Likewise, this work evidenced the seasonal variation of diversity-environment relationships in green spaces, both at the point and patch levels.

INTRODUCCIÓN

Vivimos en el antropoceno, una época marcada por cambios ambientales con efectos negativos sobre la diversidad global y el funcionamiento de los ecosistemas (Dirzo, *et al*., 2014). Dentro de este contexto, vivimos en la era de la urbanización. Para el año 2018 cerca del 55% de la población mundial se distribuía en áreas urbanas (Organización de las Naciones Unidas, 2018). Con base en las proyecciones de crecimiento en tamaño y número de las zonas urbanas, se espera que la urbanización continúe aumentando en el corto y mediano plazo.

Desde el punto de vista de la conservación y de la ecología, las zonas urbanas representan un doble reto. Primero, la urbanización actúa como un importante motor de cambio ambiental: la aparición y expansión de las zonas urbanas están estrechamente ligadas a la degradación, fragmentación y pérdida de los hábitats naturales (Grimm, 2008). Segundo, las zonas urbanas, entendidas como ecosistemas, se encuentran dominadas por el humano. Esto es, las decisiones humanas son el principal factor que influye en las condiciones ambientales de los ecosistemas urbanos (Alberti, 2008).

Estas características de las zonas urbanas tienen consecuencias sobre el estado de la diversidad local y sobre los enfoques adecuados para estudiar sus relaciones ecológicas. Por un lado, dada la transformación del hábitat asociada a la urbanización, esta última ha sido señalada como una amenaza importante a la biodiversidad (McKinney, 2002). Por otro lado, para comprender la relación entre los factores ecológicos de los ecosistemas urbanos y las especies y comunidades bióticas, se ha propuesto considerar una serie de mecanismos que pueden o no ser impulsados por las actividades humanas (Aronson *et al*., 2016).

El conocimiento sobre la biodiversidad urbana proviene en gran parte de estudios realizados en áreas verdes de distinto tipo (Gallo *et al*., 2017; Villarroya-Villalba *et al*., 2021). Esto se debe, en parte, a que las áreas verdes pueden albergar niveles de diversidad más altos que otros usos de suelo urbanos (Hayes *et al*., 2020) por lo que podrían funcionar como sitios importantes para la conservación de especies nativas dentro de las zonas urbanas (Ofori *et al*., 2018). Se sabe que los factores que pueden influir sobre la diversidad biológica al interior de las áreas verdes urbanas son múltiples y que pueden ser abióticos, bióticos y antrópicos. Por ejemplo, se sabe que la biodiversidad de estos sitios puede mostrar una relación significativa con la temperatura y precipitación, la presencia de depredadores como gatos domésticos y el uso de pesticidas como parte del mantenimiento al que están sujetos (Sims *et al*., 2008; Shwartz *et al*., 2013; Liu *et al*., 2019).

Una pregunta que se ha explorado en las áreas verdes y parches con vegetación urbanos tiene que ver con la relevancia de la matriz que los rodea (Shwartz *et al*., 2013; Zungu *et al*., 2020). Dicha pregunta se sustenta teóricamente en la siguiente premisa: la matriz alrededor de los parches de hábitat es biológicamente relevante para las especies que ocupan dichos parches (Dunford y Freemark, 2005). Para responder esta interrogante, algunos trabajos realizados han evaluado el efecto de las variables ambientales agrupándolas en factores del parche y factores de la matriz (Suarez-Rubio y Thomlinson, 2009; Rico-Silva *et al*., 2020). Entre sus resultados destacan los siguientes: 1) las variables del parche y de la matriz tienen efectos significativos sobre las especies y 2) las especies pertenecientes a distintas categorías tales como especies residentes, migratorias, sinantrópicas, anisantrópica, endémicas o exóticas responden de manera diferente a las características de la matriz (Suarez-Rubio y Thomlinson, 2009; Litteral y Wu, 2012; Rico-Silva *et al*., 2020).

Estos hallazgos podrían contribuir a comprender por qué algunas especies reducen su tamaño poblacional o desaparecen de las zonas urbanas mientras que otras logran persistir o prosperar en ellas (Blair, 1996). Pero más importante aún, al entender el papel de la matriz que rodea a las áreas verdes, se podrían diseñar estrategias de mejoramiento ambiental con el fin de mantener e incrementar la diversidad de las áreas verdes, considerando que, en la actualidad es prácticamente imposible aumentar el tamaño de las áreas verdes.

Una cuestión igual de relevante es la identificación de procesos que ocurren a escalas espaciales por debajo del parche. Dentro los estudios sobre diversidad en áreas verdes, una práctica común consiste en reunir los datos colectados en las estaciones de muestreo para obtener un único valor para cada área verde (Suarez-Rubio y Thomlinson, 2009; Malagamba-Rubio *et al*., 2013). Si bien, esto es metodológicamente correcto y permite explorar diferentes preguntas, conlleva la pérdida de información de las escalas espaciales más finas.

Estudios desarrollados con un enfoque multiescala han documentado que la ocurrencia, riqueza y abundancia de especies en parches de hábitat dentro de paisajes urbanizados dependen de características ambientales definidas a diferentes escalas espaciales

(Melles *et al*., 2003; Bhakti *et al*., 2018; Han *et al*., 2021). Los resultados de los estudios multiescala no son concluyentes respecto a qué escala espacial tiene el mayor efecto sobre los distintos parámetros de las comunidades.

Ante este panorama, esta investigación tuvo cuatro objetivos centrales: 1) recuperar la información disponible sobre el estudio de la avifauna en áreas verdes urbanas de México con el fin de establecer una línea base tanto para el presente trabajo como para futuras investigaciones; 2) estudiar la respuesta de las especies de aves a características del hábitat al interior y en las inmediaciones de las áreas verdes; 3) evaluar la relación entre características de las áreas verdes y la diversidad de aves distinguiendo los procesos que ocurren a tres escalas espaciales: a nivel de matriz, a nivel de parche y por debajo del nivel de parche; y 4) complementar el conocimiento sobre la relación entre la diversidad de aves y el ambiente evaluado a múltiples escalas mediante el uso de otras medidas de diversidad.

En ese sentido, el capítulo I de este trabajo consta de una revisión de literatura. Los resultados de la revisión comprenden una clasificación y descripción los trabajos recuperados. Como parte de los resultados también se identificaron relaciones entre la avifauna y las características de las áreas verdes que fueron consistentes entre los trabajos. La revisión permitió además identificar vacíos de información en cuanto a la cobertura geográfica de la información reunida. La revisión de literatura se publicó como parte de un libro que reúne experiencias sobre el conocimiento y conservación de fauna en ambientes modificados por el hombre en México y Chile (Nava-Díaz, 2016).

Con base en la revisión, en el capítulo II se estudiaron algunos aspectos poco explorados hasta entonces en México. Por un lado, se evaluó la influencia de la vegetación sobre la diversidad de aves, diferenciando la flora nativa de la flora exótica y por el otro lado, se estudió el efecto de la cobertura vegetal en las inmediaciones de las áreas verdes. Para cubrir parte del vacío de información detectado en el trabajo de revisión, el capítulo II se preparó con datos generados para la ciudad de Mérida, Yucatán, ubicada en una región escasamente estudiada. Los resultados del capítulo II fueron publicados en la revista "Tropical Conservation Ecology" (Nava-Díaz *et al*., 2020).

Por último, en el capítulo III se exploró la respuesta de la diversidad de aves a características de las áreas verdes definidas a tres escalas espaciales: el punto de conteo, el parche y la matriz alrededor del parche. Además de considerar la diversidad taxonómica de las áreas verdes, se trabajó con medidas de diversidad filogenética y funcional. La relación entre las variables de respuesta e independientes se analizó a nivel de área verde y a nivel de punto de conteo. Las variables se eligieron con base en lo reportado por la literatura y considerando también que puedan ser incluidas en planes de manejo. El trabajo se desarrolló en áreas verdes de la ciudad de México, una de las ciudades mexicanas mejor estudiadas lo que permitió contrastar los resultados obtenidos con la evidencia existente.

CAPÍTULO 1. Diversidad de aves en áreas verdes de zonas urbanas: una revisión para México

Resumen

En respuesta a la crisis del cambio global, particularmente a las altas tasas de urbanización, el estudio de la biodiversidad presente en zonas urbanas ha adquirido creciente relevancia. Entre los diferentes usos de suelo, las áreas verdes han recibido especial atención por ser sitios considerados potenciales refugios de la biodiversidad urbana. En esta revisión se compendiaron los trabajos sobre avifauna en áreas verdes dentro de zonas urbanas de México. Se estudió la distribución espacial de los trabajos y la proporción de las especies reportadas para las provincias biogeográficas y entidades federativas del país que cuentan con información. Se identificaron los tipos de estudios más frecuentes y se extrajeron los principales resultados. Se reunieron 42 trabajos, la mayoría realizados en la provincia del Eje Neovolcánico. Aun cuando 30 trabajos corresponden a listados de especies que describen la avifauna de uno o varios sitios, la riqueza de especies estatal o regional está poco representada en estos trabajos. Se sintetizaron los resultados de los estudios ecológicos destacando los patrones consistentes como la relación positiva entre el tamaño del área verde y la diversidad de aves y señalando también aquellos resultados discrepantes. Finalmente se plantearon preguntas de investigación que considero ineludibles.

Introducción

En las últimas décadas se han desarrollado numerosos trabajos que describen, comparan y explican los patrones de biodiversidad registrados en diversas zonas urbanas del planeta (Blair, 1999; Aronson *et al*., 2014; MacGregor-Fors *et al*., 2015) lo cual responde al efecto que éstas tienen sobre el cambio ambiental global (Grimm, 2008) y a los escenarios de urbanización proyectados para diferentes regiones del mundo (Seto *et al*., 2012). Si bien la urbanización puede implicar una pérdida de especies (McKinney, 2002), existen usos de suelo dentro de las zonas urbanas que pueden albergar niveles de riqueza relativamente altos (Blair, 1999), como las áreas verdes (Fernández-Juricic y Jokimaki, 2001). Los objetivos de esta revisión fueron: compendiar la información disponible sobre la avifauna de áreas verdes de zonas urbanas en México, determinar la riqueza de aves de México registrada en dichas áreas verdes y sintetizar los resultados de los trabajos.

Metodología

Se realizó una búsqueda con los términos “avifauna” o “ave”, “ciudad” o “urbanización” o “urbana” y “México” durante agosto del 2015 en diferentes bases de datos de publicaciones indexadas, resúmenes de congresos y portales de revistas asociaciones civiles y sociedades científicas mexicanas que realizan investigación sobre las aves. Se consultó el catálogo de tesis de la Universidad Nacional Autónoma de México. Se insertaron los términos equivalentes en inglés cuando fue necesario. En la revisión se incluyeron únicamente aquellos trabajos a nivel de comunidad que realizaron muestreos en áreas verdes dentro de zonas urbanas. Las áreas verdes fueron definidas como aquellos sitios públicos o privados arbolados de diferente uso y forma.

Los trabajos fueron clasificados en tres categorías: listados de especies, estudios ecológicos y estudios de conservación de acuerdo a lo propuesto por Ortega-Álvarez y MacGregor-Fors (2011). Para aquellos trabajos realizados en la misma zona urbana se verificó que los datos reportados fueran distintos para ser incluidos en la revisión. Se estudió la distribución espacial de los trabajos reunidos así como la riqueza de aves reportada en conjunto para cada entidad federativa y provincia biogeográfica representada en la revisión. El listado de especies de las provincias se obtuvo integrando los listados de aquellas Áreas de Importancia para la Conservación de las Aves (AICA) ubicadas en cada provincia representada cuyos límites geográficos no rebasaran los de la provincia correspondiente.

3. Resultados

Estudios de avifauna en áreas verdes urbanas

Se reunieron 42 trabajos sobre avifauna realizados en áreas verdes dentro de zonas urbanas de México. Los trabajos corresponden a 22 artículos, once tesis, ocho presentaciones en congresos y un libro, los cuales fueron publicados o presentados entre el año 2000 y 2014. Los trabajos reunidos abarcan distintos usos de suelos tales como parques, campus universitarios, jardines, solares, cementerios y jardines botánicos (Buzo-Franco *et al*., 2005;

Díaz, 2008; Parra *et al*., 2010; Domínguez *et al*., 2011; Cárdenas, 2014; Castro-Torreblanca y Blancas, 2014).

Distribución espacial de los estudios

Los trabajos identificados no están distribuidos homogéneamente en México. La mayoría de los estudios han sido realizados en la región centro del país (Fig. 1). Las entidades federativas para las que se encontró por lo menos un trabajo son: Distrito Federal (17), Estado de México (6), Puebla (6), Jalisco (6), Hidalgo (3), Veracruz (2), Chihuahua (2), Tamaulipas (1), Guerrero (1), Durango (1), Oaxaca (1), Querétaro (1), Morelos (1) y Yucatán (1). De las diecinueve provincias que comprende la regionalización biogeográfica de México solo ocho cuentan con al menos un trabajo, siendo la provincia del Eje Neovolcánico la que más trabajos presenta (Fig. 1).

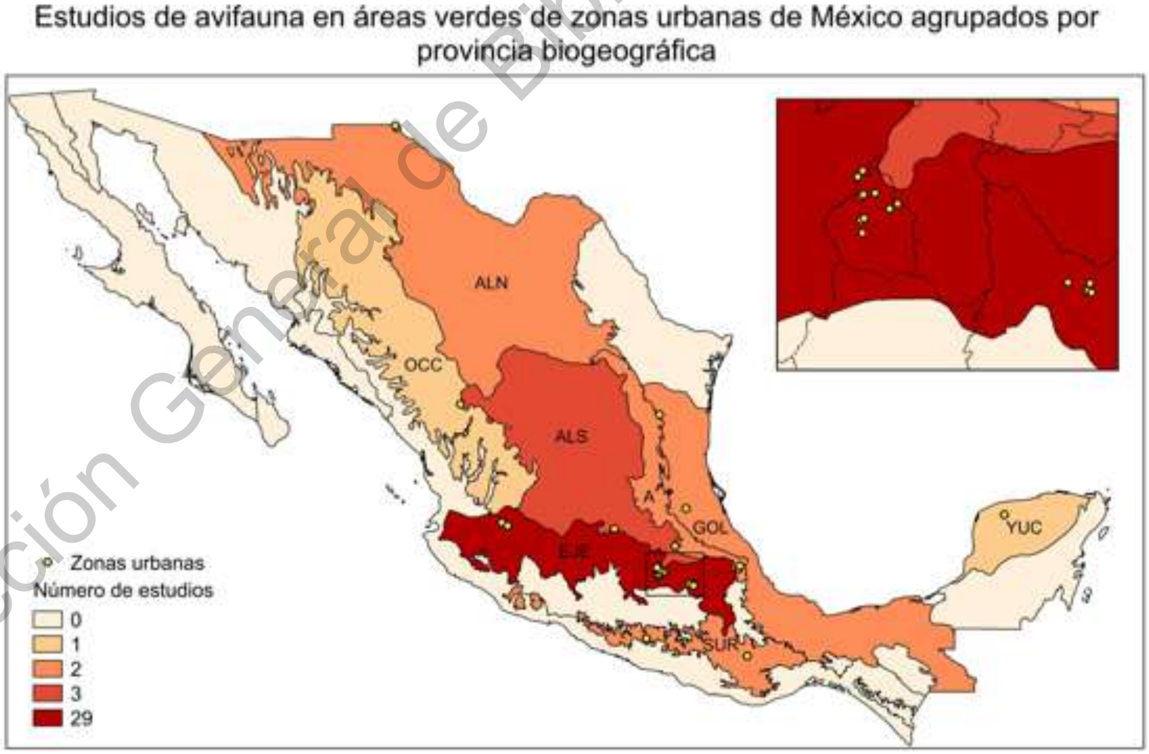

Fig. 1. Ubicación de los estudios reunidos en las provincias biogeográficas del país. Modificado de: CONABIO (1997). ALN: Altiplano del Norte, ALS: Altiplano del Sur, OCC: Sierra Madre Occidental, A: Sierra Madre Oriental, SUR: Sierra Madre del Sur, EJE: Eje Neovolcánico, GOL: Golfo de México, YUC: Yucatán.

Riqueza de aves en México reportada en los estudios

Del total de trabajos incluidos en la revisión solo se obtuvo el listado de especies para 36 de ellos. Estos trabajos reportan en conjunto un total de 469 especies pertenecientes a diecinueve órdenes y 62 familias, lo que corresponde a aproximadamente el 41% de las especies registradas para México (Navarro-Sigüenza *et al.*, 2014). De éstas, 31 especies son endémicas para México y 53 especies aparecen en alguna categoría de riesgo según la autoridad ambiental mexicana (SERMANAT, 2010). Las familias mejor representadas en orden descendiente son Parulidae, Tyrannidae, Emberizidae. La mayoría de las especies (76.7%) fue reportada en hasta siete trabajos mientras que catorce especies (2.9%) aparecieron en 27 o más listados (Tabla 1). A nivel de estado, las especies registradas en los trabajos reunidos representan desde el 6.6% hasta el 86.0% del número de especies reportadas para cada estado siendo la media y mediana 27% y 22% respectivamente (Tabla 2). El análisis de la riqueza de aves a nivel de provincia biogeográfica solo se realizó para las provincias Altiplano del Norte, Eje Neovolcánico y Golfo de México por ser las únicas con suficiente información disponible (Tabla 2).

Tabla 1. Especies más frecuentes en los 36 listados de avifauna encontrados para áreas verdes de zonas urbanas de México. Números de listados en los que aparecen (N).

N	Especies
34	*Columbina inca*
33	*Haemorhous mexicanus*
32	*Quiscalus mexicanus*
31	*Cardellina pusilla, Passer domesticus, Setophaga coronata, Spinus psaltria*
29	*Columba livia, Hirundo rustica, Molothrus aeneus, Pyrocephalus rubinus*
28	*Melozone fusca*
27	*Thryomanes bewickii, Toxostoma curvirostre*

Tipos de estudios identificados

La mayor parte de los trabajos encontrados corresponden a listados de especies (30). Solo un trabajo fue catalogado como un estudio de conservación. El resto de los trabajos fueron clasificados como estudios ecológicos (11).

3.4.1 Estudios de conservación

Reyna y colaboradores (2013) identificaron especies de aves indicadoras de la calidad ambiental de parques en la zona metropolitana y conurbada de Guadalajara. A partir de la distribución de las especies en los dieciséis parques monitoreados, obtuvieron un conjunto de ochos especies que consideraron indicadoras de ambientes perturbados y un conjunto de dieciséis especies raras. El primer grupo incluye especies como *Turdus rufopalliatus*, *Quiscalus mexicanus*, *Bubulcus ibis*, *Haemorhous mexicanus* y *Cynanthus latirostris*. En el segundo grupo se encuentran especies como *Passerina leclancherii*, *Momotus mexicanus*, *Tyto alba* y *Bombycilla cedrorum*.

Tabla 2. Representación de la riqueza de especies de las entidades federativas y provincias biogeográficas para las que se encontró al menos un trabajo. La riqueza de especies de referencia se obtuvo de Navarro-Sigüenza y colaboradores (2014) y de CONABIO (2015).

Sitio (número de trabajos)	Diversidad reportada en los trabajos incluidos en la revisión			Riqueza
	Órdenes	Familias	Especies (%)	Especies
Chihuahua (2)	16	44	162 (50.7)	319
Distrito Federal (8)	18	47	209 (86.0)	243
Durango (1)	15	43	163 (55.0)	296
Estado de México (2)	10	31	100 (22.7)	440
Guerrero (1)	8	24	76 (14.1)	539
Hidalgo (3)	14	35	106 (23.2)	456
Jalisco (6)	18	40	160 (29.1)	549
Oaxaca (1)	8	20	49 (6.6)	736
Puebla (3)	10	30	80 (13.4)	595
Querétaro (1)	6	20	39 (10.4)	375
Tamaulipas (1)	12	25	44 (8.0)	545
Veracruz (2)	18	48	265 (36.8)	719
Altiplano del Norte (2)	16	45	158 (44.1)	358
Eje Neovolcánico (23)	19	49	301 (71.1)	423
Golfo de México (2)	16	35	93 (15.6)	594

Listados de especies

Se han generado listados de especies exclusivos para un área verde (Villafranco, 2000; Rodríguez *et al*., 2010; Ruelas y Aguilar, 2010; Hernández y Peláez, 2012) o para un conjunto de áreas verdes ubicadas dentro de la misma zona urbana (González-Oreja *et al*., 2007; Martínez, 2011; Maya-Elizarrarás, 2011; Malagamba-Rubio *et al*., 2013; Resendiz *et al*., 2013; Cárdenas, 2014) siendo los primeros los más comunes y dentro de éstos, son frecuentes los que reportan la avifauna presente en campus universitarios u otras instalaciones de entidades académicas (Carbó-Ramírez y López-Ortega, 2005; MacGregor-Fors, 2005; Rodríguez, 2007; Ramírez-Albores, 2008; Jiménez y Mendoza, 2010; Pablo-López y Díaz-Porras, 2011; Hernández *et al*., 2013). Solo un trabajo ha estudiado la avifauna de áreas verdes urbanas junto con otras coberturas de suelo no urbanas dentro de un municipio (Moreno *et al*., 2015).

Algunos de los trabajos que registraron la diversidad de aves en un solo sitio evaluaron la variación espacial y temporal de ésta (Almazán-Núñez y Hinterholzer-Rodríguez, 2010; San José *et al*. 2010; Carbó-Ramírez *et al*., 2011). Las comparaciones espaciales se han realizado para sitios que difieren en las características de la vegetación presente (Carbó-Ramírez *et al*., 2011) o en el manejo que han recibido (San José *et al*. 2010) encontrándose diferencias en la composición de especies. La dinámica temporal ha sido evaluada en un mismo año (Arenas, 2004; Hernández y Peláez, 2012) o a lo largo de varios años (Ramírez-Albores, 2008). Por ejemplo, Acuña (2014) comparó la diversidad y abundancia de especies en la Facultad de Estudios Superiores Iztacala en la Ciudad de México con datos obtenidos en años previos (Duarte, 2001; Varona, 2001). Entre los cambios que reportó, está el incremento en la abundancia y frecuencia de especies como *Cynanthus latirostris*, *Psaltriparus minimus* y *Thryomanes bewickii*.

Estudios ecológicos

Los trabajos de ecología reunidos buscan explicar la variación en la diversidad de especies con base en factores como el tamaño del área verde (Buzo-Franco y Hernández-Santín, 2004), las características del hábitat (Grajales, 2009; González-Oreja et al., 2012) y del paisaje circundante (Carbó-Ramírez y Zuria, 2011), la presencia de una o varias especies urbanófilas (Ortega-Álvarez y MacGregor-Fors, 2010; Charre, 2013), la perturbación antrópica (Carbó-Ramírez y Zuria, 2011; González-Oreja *et al*., 2012), la distancia al área natural más próxima (Charre, 2013). El efecto de las variables se ha modelado para el conjunto de especies registradas (Grajales, 2009) o para un subconjunto de ellas (Ortega-Álvarez y MacGregor-Fors, 2010; Charre, 2013).

Algunos de los resultados presentados por los trabajos reunidos sugieren patrones consistentes en las diferentes zonas urbanas estudiadas como la relación positiva entre el tamaño del área verde y la riqueza de especies (Buzo-Franco y Hernández-Santín, 2004; Carbó-Ramírez y Zuria, 2011; González-Oreja *et al*., 2012; Charre, 2013) aunque en algunos casos, la relación no ha sido significativa (MacGregor-Fors y Ortega-Álvarez, 2011). En otros casos, los resultados obtenidos por diferentes estudios son desiguales. Por ejemplo, la distancia entre las áreas verdes y el área natural o preservada más cercana muestra un efecto negativo sobre la riqueza de especies (Grajales, 2009), un efecto nulo sobre el número de individuos de especies migratorias (Charre, 2013) o un efecto positivo sobre la riqueza de especies migratorias (Charre *et al*., 2013).

La caracterización del hábitat es común en los estudios ecológicos compilados. Ésta se basa frecuentemente en atributos de la vegetación (MacGregor-Fors, 2008; Grajales, 2009; Malagamba-Rubio *et al*., 2013) aunque también ha considerado otras variables relacionadas con la estructura urbana del sitio (Ortega-Álvarez y MacGregor-Fors, 2009). Entre los atributos de la vegetación incluidos en los análisis, la riqueza de aves muestra una relación positiva pero no significativa con la riqueza de especies vegetales, sean arborescentes o arbustivas, (MacGregor-Fors, 2008; Grajales, 2009; MacGregor-Fors y Ortega-Álvarez, 2011) y una relación positiva significativa con el diámetro a la altura del pecho (Malagamba-Rubio *et al*., 2013) y la cobertura del follaje arbóreo (MacGregor-Fors, 2008). Es importante mencionar que los trabajos que analizaron la riqueza y abundancia de grupos particulares de aves reportan relaciones especie-dependientes con algunos atributos de la vegetación. Por ejemplo, Ortega-Álvarez y MacGregor-Fors (2010) encontraron que la abundancia de *Spinus psaltria* y *Pheucticus melanocephalus* se relaciona fuertemente con la densidad de árboles mientas que la abundancia de *Turdus migratorius*, *Icterus abeillei* y *Amazilia beryllina* estn asociada a la altura de los árboles.

En cuanto a las características del hábitat relativas al grado de urbanización, se han reportado valores más altos de dominancia para aves insectívoras migratorias en los parques menos urbanizados (Charre, 2013) y se ha identificado un grupo de especies asociadas a áreas verdes con una proporción de suelo impermeable alta para las temporadas reproductiva y de migración (Charre *et al*., 2013). También se ha descrito una relación negativa entre la cobertura de suelo desnudo y la riqueza de especies (Malagamba-Rubio *et al*., 2013) la cual puede variar si se consideran diferentes grupos de especies (MacGregor-Fors y Ortega-Álvarez, 2011).

Las respuestas de las comunidades a factores del paisaje han sido escasamente estudiadas. Carbó-Ramírez y Zuria (2011) incluyeron en sus análisis la proporción de suelo cubierto por áreas verdes, caminos pavimentados y edificios en 100 y 200 m a la redonda de cada sitio de muestreo. Los modelos que obtuvieron para la comunidad de verano y de invierno incluyen como variables explicativas de la riqueza el porcentaje de suelo cubierto por edificios y por caminos pavimentados respectivamente.

La evidencia generada señala que las perturbaciones antrópicas pueden afectar de manera negativa la riqueza y composición de las comunidades de aves en las áreas verdes (González-Oreja *et al*., 2012; Charre, 2013) y la abundancia de especies individuales (Ortega-Álvarez y MacGregor-Fors, 2010) aunque en ocasiones pueden no tener efecto alguno (Carbó-Ramírez y Zuria, 2011). En un trabajo pionero, González-Oreja y colaboradores (2012) investigaron el papel del ruido en el anidamiento de las comunidades de aves en áreas verdes. Los modelos finales que obtuvieron muestran que el anidamiento de los ensambles de aves es explicado por el tamaño del sitio junto con los niveles de ruido.

Conclusiones

Los resultados de esta revisión muestran que si bien son numerosos los trabajos que han investigado la diversidad de aves en áreas verdes para diferentes zonas urbanas de México, los esfuerzos se han dirigido en su mayoría a describir las avifaunas ahí presentes. Esta revisión señala también que la cobertura espacial de los trabajos es limitada, desarrollándose gran parte de éstos en la provincia biogeográfica del Eje Neovolcánico, en particular en la Ciudad de México. Si bien el total de especies de aves reportadas en los trabajos compendiados equivale al 41% de las especies reportadas para México, esta proporción disminuye importantemente si se calcula para las entidades federativas y provincias biogeográficas para las que se obtuvo información.

Esta revisión confirma que la cobertura vegetal al interior de las áreas verdes tiene un efecto significativo sobre la riqueza de aves. Los trabajos reunidos permiten señalar un único patrón consistente, áreas verdes de mayor tamaño presentarán valores más altos de riqueza de especies. La diversidad de aves en las áreas verdes es afectada por factores a nivel local y de paisaje. Entre los primeros, algunos atributos de la vegetación, así como las perturbaciones antrópicas, en particular el ruido, pueden ser factores que afectan de manera significativa la composición de las comunidades en las áreas verdes.

Perspectivas

México es un país con una alta riqueza de avifauna y una alta proporción de endemismos (Navarro-Sigüenza *et al*., 2014) y es también un país que ha experimentado en las últimas décadas un incremento en el número y extensión de sus zonas urbanas (Almejo *et al*., 2014). Dado que la urbanización supone una amenaza para la conservación de la biodiversidad (McKinney, 2002), se requiere de información que permita tanto valorar el estado actual de la biodiversidad en las zonas urbanas del país como conocer los mecanismos que la afectan.

Varios de los trabajos revisados señalan una relación entre algunos atributos de la cobertura vegetal y la diversidad de aves como se ha reportado para otras zonas urbanas (Barth *et al*., 2015). Sin embargo, poco se han explorado los mecanismos que subyacen a las relaciones reportadas. Más allá de evaluar la diversidad de especies vegetales *per se* propongo que se estudien los recursos que proporcionan las distintas especies vegetales como se ha hecho en otras áreas urbanas (Davis *et al*., 2015). Considerando que la vegetación de muchas áreas verdes y ciudades está dominada por especies exóticas como eucaliptos (*Eucalyptus* spp.), casuarinas (*Casuarina equisetifolia*) o pirules (*Schinus molle*) (Grajales, 2009; Charre *et al*., 20013) y que las aves pueden usar preferentemente estas especies sobre las nativas (MacGregor-Fors, 2008; Charre, 2013) recomiendo que se estudie explícitamente el valor que las especies vegetales exóticas tienen para la diversidad de aves en las áreas verdes.

Adicionalmente, sugiero que las investigaciones se desarrollen dentro del marco de la Ecología del Paisaje ya que las zonas urbanas son espacialmente heterogéneas. Al respecto considero relevante explorar el efecto de la conectividad de las áreas verdes sobre la relación tamaño-riqueza observada en éstas (Martensen *et al*., 2008). Por último, resalto la necesidad de determinar si las áreas verdes dentro de las zonas urbanas funcionan como trampas ecológicas para las especies que ahí habitan o si verdaderamente representan un refugio para la diversidad de aves (Leston y Rodewald, 2006).

CAPÍTULO 2. Drivers of functional composition of bird assemblages in green spaces of a Neotropical city: a case study from Merida, Mexico

Abstract

Given current urbanization trends, understanding the factors that affect local biodiversity is paramount for designing sound management practices. Existing evidence suggests that the assembly of urban communities is influenced by the environmental filtering of organisms based on their traits. Here, we investigate how environmental characteristics including isolation measurements affect the functional composition of avian assemblages in green spaces of Merida, Mexico, a Neotropical city. We sampled 22 sites, analyzed point-count data collected during fall migration, and characterized the habitat with regard to floristic and structural vegetation attributes, vegetation cover within green spaces, urban infrastructure and isolation. We assessed the relationship between habitat descriptors and bird functional traits using RLQ and fourth-corner tests and compared trait-environment associations between resident and wintering species. Our results showed that functional composition of resident bird assemblages was linked to the environmental characteristics of the site, while the functional composition of wintering species was not. In particular, the degree of isolation revealed to be an important determinant of trait composition. Plant species richness, particularly native tree and shrub species, were critical for the functional composition of resident birds in green spaces. Our findings suggested shifts in body mass from less to more isolated green spaces. Specifically, we observed that large-bodied species predominated in isolated green spaces. This information is useful given the predicted increases in habitat isolation and transformation of green spaces due to urbanization.

Introduction

Urban ecosystems are complex dynamic systems where humans are the dominant driving force (Alberti, 2008). Major human-induced transformations within urban areas include the clearing of vegetation, the introduction of non-native plant species, the installation of artificial structures, and the alteration of the quality and quantity of disturbances (Niemela, 2011; Parris, 2016) which can have significant effects on the spatial distribution of urban fauna (Fernandez-Juricic, 2002; González-Oreja, De La Fuente-Díaz-Ordaz, Hernández-Santín, Bonanche-Regidor, & Buzo-Franco, 2012; Ortega-Álvarez & MacGregor-Fors, 2010; White, Antos, Fitzsimons, & Palmer, 2005). In the face of global urbanization trends (Fragkias, Günerlap, Seto, & Goodness, 2013), understanding the factors that drive biodiversity patterns in urban areas has become paramount for both environmental science and policy.

Birds stand as one of the most common models to study wildlife responses to urbanization (Murgui & Hedblom, 2017). The majority of urban bird studies are conducted within vegetated green spaces due to their biodiversity conservation potential (Gallo, Fidino, Lehrer, & Magle, 2017). Urban green spaces can encompass sites that resemble natural habitats to a varying extent: from remnants of the local original vegetation to areas exclusively intended for human use. As a consequence, green spaces can markedly differ in size, can be subject to contrasting management practices, and can be used in distinct ways by visitors, all this variation affecting the conservation value of urban green spaces (Carbó-Ramirez & Zuria, 2011; Fernandez-Juricic, 2002; Tryjanowski *et al*., 2017).

Current understanding of the influence of green spaces characteristics on species richness and abundance is deep (Nielsen, van den Bosch, Maruthaveeran, & van den Bosch, 2014). However, it is equally important to gain an insight into the functional component of green spaces' biodiversity (Pavoine & Bonsall, 2011). Furthermore, it has been proposed that the assembly of urban communities is determined, in part, by the interaction of environmental filters and species traits (Aronson *et al*., 2016). An increasing number of publications have documented the influence of bird species' traits on their susceptibility to urbanization. Commonly assessed traits include trophic guild, migratory status and body mass (Lees & Moura, 2017) while some authors have broadened the set of traits analyzed considering characteristics such as adult survival rate or innovative behavior (Meffert & Dziock, 2013). Recent works have quantified the functional diversity of urban avifauna through the use of indices (Morelli *et al*., 2017; Schütz & Schulze, 2015). While such studies contribute to understand the effect of the filters on the distribution of traits, they do not allow to identify habitat associations with traits. We expect that if environmental factors prevent or favor the establishment of birds in green areas based on their traits, the distribution of species in surveyed green spaces will be heterogeneous, with species holding similar traits responding in a common fashion to habitat characteristics (Kraft *et al*., 2015).

Understanding how species' traits are related to environmental characteristics of urban or urbanizing sites is paramount, especially in those areas experiencing or projected to experience elevated urbanization rates such as Mexico. Despite urban bird ecology in Mexico has experienced a rapid growth in recent years (Marzluff, 2017) most of the published information refers to urban areas within the Trans-Mexican Volcanic Belt (Nava-Díaz, 2016), while other important biogeographic regions remain unexplored not to mention. Though previous studies have assessed the responses of resident and migratory species at their breeding grounds (Huste & Boulinier, 2011), studies in their wintering grounds are uncommon (but see Wolf, Degregorio, Rodriguez-Cruz, Mulero-Oliveras, & Sperry, 2018). Furthermore, functional traits information has been missing in urban bird ecology research in Mexico, despite it can contribute to disentangle the relationship between avian communities and urban-related habitat transformations (Silva, Sepulveda, & Barbosa, 2016). In order to fill an important information gap, we explored trait composition determinants during autumn migration in a Neotropical city. More precisely, this study was aimed: (1) to explore how species and trait composition change in urban green spaces, (2) to identify traits that predict species response to habitat characteristics within green spaces, and (3) to compare trait-environment associations between resident and wintering species.

Methods

Study area

Fieldwork was carried out in Merida (approximately 20.9º N, 89.6º W, 15 m a.s.l.), the main city of the Yucatan Peninsula (YP), southeastern Mexico with more than 1.1 million people (Consejo Nacional de Población [CONAPO], 2016). YP is one of the Mexican biogeographic regions with highest levels of bird species richness (Navarro-Sigüenza *et al*., 2014) and it holds considerable importance for wintering and transient Nearctic-Neotropical species (Calmé, MacKinnon, Leyequién, & Escalona-Segura, 2015). The area was originally covered by seasonally dry tropical forest characterized by a dry season that may last between 7 and 8 months (Torrescano-Valle & Folan, 2015). Currently, more than 25% of the native species that conform the urban flora belong to the Fabaceae, Euphorbiaceae, and Poaceae families while common exotic species include Flamboyant (*Delonix regia*), Golden rain tree (*Cassia fistula*), and Indian almond (*Terminalia catappa*) (Peraza-Contreras, 2011).

A total of twenty-two green spaces were surveyed within Merida Municipality limits (Fig. 1). Herein, we use green spaces to refer to urban open spaces dominated by trees and shrubs that are used by humans for several purposes such as recreation, exercise, education, or others, and to which access can be unrestricted or restricted. For this work we surveyed botanical gardens, public parks, an archaeological site, a zoo, a sport complex, and a reforested area within an industrial plant (Fig. 2). Green space size ranged from 0.5 to 39.1 ha and distance to the nearest native vegetation patch ranged from 101 to 5685 m. Some of the surveyed sites harbor artificial waterbodies and were included in the sample to acknowledge the importance of these habitats for local bird diversity given that superficial waterbodies are scarce in YP (Torrescano-Valle & Folan, 2015).

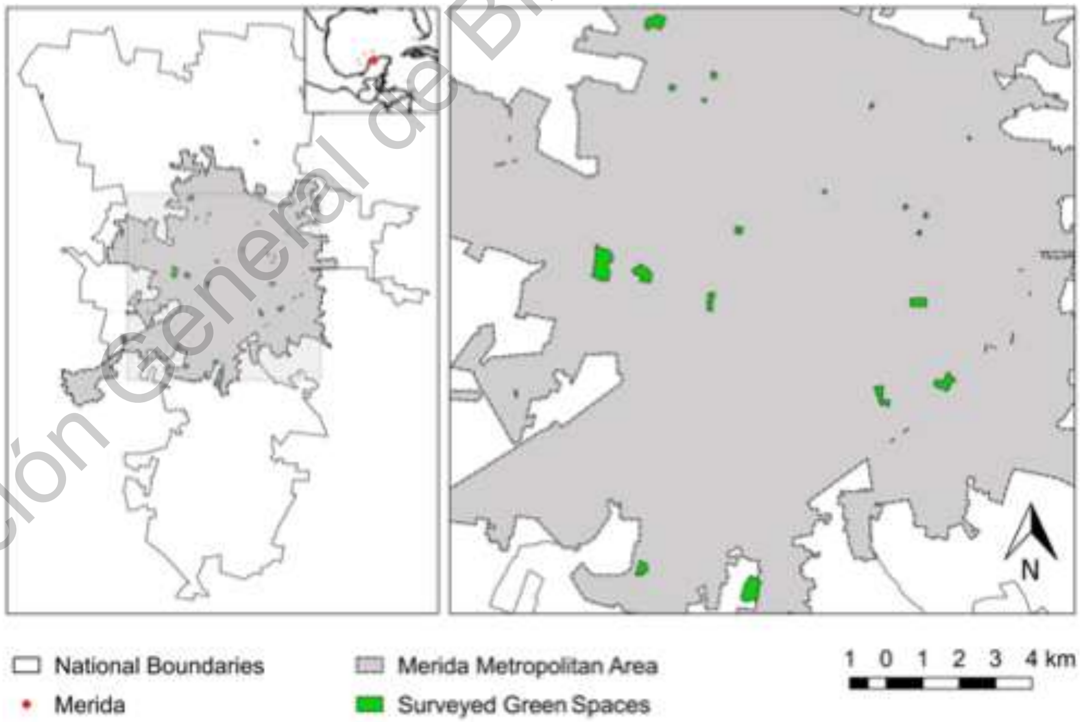

Fig 1. Left: Study area (represented as the red dot on the Mexico inset map) and overview of the 22 surveyed green spaces within Merida Municipality, Yucatan, Mexico. Right: Zoom to the shadowed area on left.

Fig. 2. Surveyed green spaces in the city of Merida encompass a wide variety of habitas.

Bird surveys

Bird surveys were conducted during the fall of 2016 from September 27th to November 15th. Birds were surveyed using 5 min fixed-width point counts (25 m) separated by at least 150 m. We chose a 25-m radius to increase the probability that all individuals would be detected in all the surveyed habitats (Hutto, 1986). The number of sampling points ranged from one to eight and corresponded to green space size. Counts were made during the first four hours after sunrise under suitable weather conditions (Ralph *et al*., 1996). Each point was visited thrice and bird data were collected by a single observer (RND). All detected birds were included in the avifauna description except *Chaetura vauxi* and *Stelgidopteryx serripennis*, since these species were flying over the plots and hence, were unlikely to be using the habitat within the plot (Gates, 1997). Statistical analyses were performed with landbirds only since we did not measure the main habitat features that influence the distribution of aquatic birds (Rosa, Palmeirim, & Moreira, 2003). Raptors were not included in statistical analysis because the count method is not suitable for estimating their numbers (Fuller & Mosher, 1981). For each sampling point, we pooled data from all three visits to get cumulative lists of detected species and generate a species presence/absence table. To determine if our survey effort was enough to provide a representative sample of the bird community in the time surveyed, we computed the non-parametric incidence based estimator Jacknife 1 (González-Oreja *et al*., 2010) using EstimateS ver. 9.1.0 (Colwell, 2013).

Habitat characterization

We measured nine variables to evaluate the habitat using 25-m-radius circular plots centered on each bird sampling point (Table 1). We considered five classes of environmental variables that could influence the distribution of birds in green spaces. Vegetation composition was

Table 1. Descriptive statistics of environmental variables recorded in green spaces of Merida, Yucatan. Shown are variables' names, abbreviation, description, and the corresponding predictor set.

Predictor set	Variable Code	Description (units)	Range	Median	Mean ± se
Vegetation composition	TreeN	Native tree species richness	1 - 14	5	5.2 ± 3.6
	TreeE	Exotic tree species richness	0 - 100	79.2	63.4 ± 39.5
	ShrubN	Native shrub species richness	0 - 16	1	3.5 ± 4.1
	ShrubE	Exotic shrub species richness	0 - 100	73.2	64.6 ± 40.8
Vegetation structure	TreeHe	Maximum tree height (m)	5.6 – 13.2	8.6	8.8 ± 2.0
	TreeAr	Maximum tree basal area (m^2)	0.02 – 3.58	0.13	0.36 ± 0.77
	ShrubHe	Maximum bush height (m)	0.5 - 5.0	3.9	3.5 ± 1.2
	ShrubAr	Maximum bush basal area (cm^2)	1.5 – 652.8	59.3	138.9 ± 176.1
Vegetation cover	Veg1	Class 1 vegetation cover (ha)	0.03 – 4.47	0.40	0.93 ± 1.25
	Veg2	Class 2 vegetation cover (ha)	0.05 – 5.79	0.44	0.97 ± 0.14
	Veg3	Class 3 vegetation cover (ha)	0.07 – 6.55	0.46	1.14 ± 0.16
	Veg4	Class 4 vegetation cover (ha)	0.01 – 9.19	0.53	1.44 ± 2.37
	Veg5	Class 5 vegetation cover (ha)	0 – 16.8	0.19	1.72 ± 3.86

	Veg6	Class 6 vegetation cover (ha)	0 – 1.83	0	0.11 ± 0.39
Urban infrastructure	Poles	Number of poles	0 - 10	4	4.1 ± 3.0
Patch extra predictors	Size	Green space size (ha)	0.5 - 39.1	2.7	7.8 ± 10.0
	Distance	Distance to the closest native vegetation patch (m)	101.5 – 5685.0	2485.4	2670.5 ± 1594.0
	SVeg100	Scattered vegetation cover in a 100-m width buffer around each green space (ha)	0.06 – 3.95	1.22	1.52 ± 1.14
	DVeg100	Dense vegetation cover in a 100-m width buffer around each green space (ha)	0 – 11.83	0.35	2.00 ± 3.57

evaluated by: native tree species richness, exotic tree species richness, native shrub species richness, and exotic shrub species richness. Vegetation structure was assessed by: maximum tree height, maximum basal area, maximum bush height, and maximum bush basal area. To quantify the extent of the urban infrastructure in green spaces, we counted the number of poles.

Vegetation coverage within each green space was estimated using the Soil Adjusted Vegetation Index (SAVI), which is a modified version of the Normalized Vegetation Index (NDVI) (Huete, 1988). NDVI and SAVI are strongly correlated to several vegetation parameters including vegetation density and percent green vegetation cover (Huete, 1988; Purevdoj, Tateishi, Ishiyama & Honda, 1998). In the present study, we employed SAVI because it minimizes errors due to soil substrate optical properties (Huete, 1988). Using a Copernicus Sentinel-2 satellite image and an open source geographic information system (QGIS Development Team, 2020), we computed SAVI using the formula:

$$\text{SAVI} = [(\text{NIR} - \text{red}) / (\text{NIR} + \text{red} + L)] * (1 + L)$$

Where
NIR = Near-infrared band
Red = Red band
L = 0.5

The quality of images acquired during the bird sampling season was low due to cloud cover so we used a satellite image acquired on January 25th 2017 (cloud cover percentage = 0.0 %). The resulting SAVI layer had a 10-meter spatial resolution. SAVI values were classified in twelve classes ranging from one (built environment) to 11 (very dense vegetation) and zero values represent water. High-resolution Google Maps images were used to assign SAVI classes that represent vegetation, considering from very scattered vegetation to highly dense vegetation. Once pixels were classified in one of these twelve classes, we obtained the total number of pixels for each class. Then, total counts were used to estimate vegetation cover.

Green space area can affect the probability of occupation of species in different fashion (Roberts & King, 2017), so we estimated green space size to introduce it in the models. Similarly, isolation of green spaces can influence the composition structure of urban bird communities (Charre, Hurtado, Neve, Ponce-Mendoza, & Corcuera, 2013; Fernandez-Juricic, 2000). In this study, we used two alternative approaches to quantify green space. For the first one, we calculated the Euclidean distance from each green space to the closest continuous native vegetation patch. The second approach considered the fact that vegetation cover is not homogeneously distributed through the city and that vegetation cover adjacent to green spaces can influence bird richness and abundance (Shanahan, Miller, Possingham & Fuller, 2011). Therefore, we calculated the extent of vegetation cover in a 100-m width buffer around each green space. Estimations of vegetation cover adjacent to green spaces did not include estimations of vegetation cover within each green space. For this purpose, we employed the same SAVI classes used to identify vegetation cover within green spaces but in the case of vegetation cover adjacent to green spaces, vegetation-related SAVI classes were grouped in two broad categories: scattered and dense vegetation

Bird trait data

All species in our database were characterized based on three functional traits: diet, foraging strata, and body mass. Diet was expressed as the percentage use of each of the 10 food items categories considered (Table 2). Percentages of diet composition for each species sum to 100. Foraging strata trait was expressed as the estimated percentage use of each one of the seven strata considered. The values of the seven strata sum to 100. Functional traits were sourced from Wilman *et al*. (2014) and Del Hoyo, Elliot, Sargatal, Christie, and de Juana (2018). Additionally, residence status in the area was determined based on digital species distribution maps (BirdLife International, 2018) and considering four categories: residents, wintering, transient, and wintering/transient species.

Data analyses

Here we investigate the effect of potentially influential environmental factors on the functional composition of avian assemblages of urban green spaces through RLQ and fourth-

Table 2. Bird traits used in this study.

Traits	Trait code
Diet composition	
Invertebrates-general (%)	Invertebrate
Mammals, Birds (%)	Endotherms
Reptiles, snakes, amphibians, salamanders (%)	Ectotherms
Fish (%)	Fish
Vertebrates-general or unknown (%)	Vertebrates
Scavenge, garbage, offal, carcasses, trawlers, carrion (%)	Scavenge
Fruit, drupes (%)	Fruits
Nectar, pollen, plant exudates, gums (%)	Nectar
Seed, maize, nuts, spores, wheat, grains (%)	Seeds
Other plant material, Grass, ground vegetation, seedlings, weeds, lichen… (%)	Plant Material
Foraging stratum	
Foraging below the water surfaces (%)	Below Water Surface
Foraging on or just (<5 inches) below water surface (%)	Around Water Surface
Foraging on ground (%)	Ground
Foraging below 2m in understory in forest, forest edges, bushes or shrubs (%)	Understory
Foraging in mid to high levels in trees or high bushes (2m upward), but below canopy (%)	Mid-high
Foraging in or just above (from) tree canopy (%)	Canopy
Foraging well above vegetation or any structures (%)	Aerial
Others	
Body mass (g)	Mass

corner tests, which allow to analyze trait-environment relationships (Dray *et al*., 2014). RLQ is a three-table ordination aimed to identify the main co-structures between an environmental table (R) and a trait table (Q) with the constriction of a species table (L). On the other hand, the fourth-corner approach quantifies and tests the significance of bivariate associations between traits and environmental variables (Dray *et al*., 2014). RLQ combines three separate ordinations which summarize the main structures of each table.

In this way, RLQ relate species traits and environmental variables considering a sites-by-species table (ter Braak, Cormont, & Dray, 2012). In this study, we employed a binary species table (presence or absence). For the independent ordinations, we followed Borcard, Gillet, and Legendre (2018) to choose the ordination method based on the type of the variables and to assign rows and columns weights. To test the significance of the association between the environmental and trait tables several permutation models have been proposed (Thioulouse *et al*., 2018; see details of permutation models in Borcard *et al*., 2018). We used a single global test that consists of two independent models whose null hypotheses are: species compositions in the sites are not related to environmental conditions of the sites (Model 2) and species distribute according to their environmental preferences but irrespective of their traits (Model 4) (Borcard *et al*., 2018). The maximum p-value of both permutation tests becomes de overall p-value to attain a correct Type 1 error (ter Braak *et al*., 2012).

Measured environmental variables describe broad habitat features, so we grouped them in distinctive sets: *vegetation composition* (4), *vegetation structure* (4), *ground cover types* (4), and *anthropic disturbance* (4). The fifth set, *patch descriptors* (2) includes green space size and its proximity to native vegetation patches. We performed an RLQ analysis including all the predictor sets (that total seventeen variables). Additionally, we performed alternative RLQ analyses excluding one of the sets in each ordination since some variables can introduce noise and reduce significance. These RLQ ordinations with reduced environmental tables were obtained to explore the contribution of broad habitat characteristics in determining the functional composition of these bird assemblages. The sum of the correlation L metric of the first two RLQ axes was used as an indicator of the goodness-of-fit of the RLQ ordination (Bernhardt-Romermann *et al*., 2008). This metric serves to compare the correlation between the trait-based species scores and the environmental-based site scores generated by the RLQ ordination and the correlation of the sites and species scores of the separate ordination of the species table (Thioulouse *et al*., 2018). The ordinations with better fit are reported and used in the species grouping (further details afterwards), and displayed in the plots.

The fourth-corner analysis tests the relationships between species traits and environmental variables, one at each time (Thioulouse *et al*., 2018). In this study, we employed jointly fourth-corner analysis and output RLQ axes, which can be interpreted as either environmental gradients or trait syndromes (Dray *et al*., 2014). Since multiple tests are performed, p-values need to be corrected. We set to 9,999 the number of permutations and used the false discovery rate method (FDR) to adjust p-values in order to avoid Type 1 error (see details in Thioulouse *et al*., 2018). We decided to report results for which p-value < 0.10 to increase the power of the test given the small sample size (Zar, 2014) and in order to detect likely associations.

In order to distinguish groups of species that share traits and respond in similar ways to environmental characteristics, we grouped species based on their resulting RLQ scores using Ward's hierarchical clustering. To determine the optimal number of groups, we considered Calinski-Harabasz index (Borcard *et al*., 2018). Only those species that occurred in three or more green spaces were included in the analyses to reduce the disproportionately large effects of rare species (Legendre & Gallagher, 2001). We investigated patterns of response of two avian categories: resident and wintering species so we performed RLQ analysis separately for these subsets of species. The experimental unit in all the multivariate analyses was the green space, hence, data from the sampling points were pooled for each green space. All statistical analyses were performed with R (R Core Team, 2018), applying functions from vegan package (Oksanen *et al*., 2019), and ade4 package (Dray & Dufour, 2007) and found in Kleyer *et al*. (2012).

Results

Surveyed green spaces were environmentally heterogeneous according to the measured variables (Table 1; Fig. 2). Some sites harbored exclusively exotic or native plant species, but green spaces with a predominance of native species were majority (~65%). Considering vegetation structure, tree stratum in green spaces tended to be less than ten meters tall, while shrub stratum height was more evenly distributed, and tree basal area and shrub area were skewed towards low values. Green spaces ranged in size from 0.5 to 39 ha but most of them (n= 16) were less than 10 ha, and they were scattered through the city, so the distance to native vegetation patches varied from *circa* 100 m to more than 5000 m. Vegetation cover within and around green spaces was represented by six classes of SAVI values. Within green spaces, vegetation covered from 51 % to 98 % of their area while vegetation cover in the vicinity of green spaces ranged from 1.8 % to 51.4 %.

Avifauna of green spaces in Merida

A total of 87 species from 16 orders and 32 families were detected in green spaces of Merida (Appendix 1). Species richness per green space ranged from nine to 43. Results of the richness estimation indicate that our sample captured 78.4 % of the species present in the surveyed space and time. There was a clear predominance of resident birds over migratory ones (Table 3). The most abundant species were Great-tailed Grackle (*Quiscalus mexicanus*) and White-winged Dove *(Zenaida asiatica*). Six species were widespread occurring in more than 80 % of the green spaces while thirty-four species such as Masked Tityra (*Tityra semifasciata*) or Vermilion Flycatcher (*Pyrocephalus rubinus*) were detected in just one green space. Aquatic birds, such as Blue-winged Teal (*Spatula discors*) or Little Blue Heron (*Egretta caerulea*), were present in only three green spaces and comprised seventeen species including migratory ones. We recorded three species endemic to the YP (Calmé *et al*., 2015): Yucatan Woodpecker (*Melanerpes pygmaeus*), Yucatan Jay (*Cyanocorax yucatanicus*), and Orange Oriole (*Icterus auratus*) and two exotic species in the area: Rock Pigeon (*Columba livia*) and Eurasian Collared-Dove (*Streptopelia decaocto*).

RLQ ordination for birds of urban green spaces

The correlation L score indicated that the optimal environmental set for resident and wintering species were similar (Table 4). The environmental table used in the RLQ analysis of resident birds included the *vegetation composition*, *vegetation structure*, *urban infrastructure, and patch extra descriptor sets.*

Table 3. Species richness, abundance and estimated biomass of species of different categories recorded in green spaces of Merida, Yucatan.

	No. species	No. individuals	Total biomass (kg)
Residence status			
Resident	58	1214	158.90
Wintering	24	145	25.6
Transient	4	57	9.12
Wintering/transient	1	12	0.11
Diet			
FruiNect	8	80	3.41
Invertebrate	44	679	63.88
Omnivore	16	279	18.57
PlantSeed	11	350	62.05
VertFishScav	8	40	45.82
Other categories			
Endemic	3	26	1.51
Exotic	2	38	8.12

For wintering species, the environmental table consisted of the same descriptor sets except for urban infrastructure. The first two axes accounted for most of the variability explained by the separate ordinations (92% and 81% for resident and wintering species, respectively), so the covariance between environmental variables and species traits was well described by RLQ analysis. Considering the first RLQ axis, variability was better captured for the environmental table (95% for resident birds, 92% for wintering birds) than for the trait table (63% for resident birds, 52% for wintering birds).

Drivers of functional composition: Resident birds

RLQ axis 1 extracted 87.5% of the co-inertia and it defined a gradient of green space isolation, driven mainly by decreases in the amount of dense vegetation and increases in the distance to native vegetation patches. (Fig. 3a). The number of plant species, especially native species of both trees and shrubs proved to be another relevant driver along RLQ axis and it showed an opposite association with isolation (Fig. 3b). The ordinations indicated that large-bodied species were common in sites far from native vegetation patches but surrounded by dense vegetation, while small-bodied species were present in green spaces that occupy an intermediate position along this axis.

Table 4. Summary of RLQ ordinations for resident and wintering species with different sets of explanatory variables.

			Coinertia (%)		Variability explained (%)	Global testing p-value	
	Sets of explanatory variables	Correlation L metric	R	Q	Axis 1, 2	Model 2	Model 4
Resident species	comp + stru + infr + patc	0.393	95.5	63.7	87.5, 5.1	0.0001	0.037
	comp + stru + patc	0.390	95.4	62.7	88.4, 5.3	0.0002	0.044
	comp + stru + cov + infr + patc	0.364	86.1	63.8	87.5, 4.5	0.001	0.073
	comp + cov + infr + patc	0.363	84.0	64.3	90.1, 4.7	0.001	0.072
	comp + stru + cov + patc	0.363	86.3	63.0	88.5, 4.6	0.001	0.082
Wintering species	comp + stru + patc	0.928	92.9	81.8	69.3, 12.4	0.122	0.764
	comp + stru + infr + patc	0.916	92.6	52.5	65.1, 17.1	0.127	0.758
	comp + stru + cov + patc	0.869	86.0	62.2	65.2, 19.3	0.096	0.559
	comp + cov + patc	0.869	88.0	62.0	67.9, 21.6	0.049	0.465
	comp + cov + infr + patc	0.853	87.0	63.0	64.9, 21.4	0.051	0.465

Ranking of RLQ ordinations was based on correlation L metric. Only the 5 best ranked ordinations for each group of species are listed. Variance captured by separate ordinations represents the maximum value, to which variance captured by corresponding RLQ axes is compared (coinertia R and coinertia Q). Percentage of variability captured by RLQ axis 1 and 2 is shown together with P-values obtained for each ordination. Abbreviation for sets of variables are (numbers in brackets correspond to the number of variables in the set): comp, vegetation composition (4); stru, vegetation structure (4); cove, vegetation cover (6); infr, urban infrastructure (1); patc, patch extra descriptors (4).

Furthermore, canopy-forager species were present in green spaces surrounded by relatively large amounts of dense vegetation and where native flora predominated (Fig. 3c). This includes species such as Scrub Euphonia (*Euphonia affinis*) or Green Jay (*Cyanocorax yncas*) (Fig. 3d). On the contrary, ground-foraging species whose diet is dominated by seed and plant material became more common in sites of reduced native flora richness. RLQ axis 2 separated green spaces based on the number of exotic tree species but it only extracted 5.1% of the co-inertia so we do not consider it for further discussion.

Global testing showed that the link between traits and environment was significant for resident birds (Model 2 *p*-value: 0.0001; Model 4 *p*-value: 0.03) and this finding was supported by fourth-corner tests on RLQ axes. The joint approach of RLQ analysis and fourth-corner tests indicated that numerous environmental factors of green spaces were related to the functional composition of bird communities (Figure 3b). In particular, the amount of dense vegetation adjacent to green space showed the highest influence on trait composition. Regarding species' traits, the association of body mass and aerial-foraging strategy with axis 1 was marginally significant (Figure 3c). There was no evidence for significant associations for RLQ axis 2. Results from the joint approach of RLQ and fourth-corner analysis are summarized in the ordinations plots (Figure 3b and 3c).

Drivers of functional composition: Wintering birds

RLQ axis 1 (69.3% of co-inertia) separated less isolated green spaces harboring a large number of shrub species, both native and exotic ones, from those green spaces with impoverished shrub richness, but with an elevated number of exotic tree species and far from native vegetation patches (Figure 3a). Body size was related to this axis together with the aerial-foraging strategy. Other traits, such as understorey-foraging and midhigh-foraging were not clearly associated with any particular characteristic of green spaces.

Global RLQ testing did not support a significant association between traits and environment (Model 2 *p*-value: 0.12; Model 4 *p*-value: 0.76). However, fourth-corner tests indicated few marginally significant associations (Figures 3b and 3c): dense vegetation extent and native shrubs richness were the characteristics of green spaces associated with trait composition, whereas body mass and the aerial-foraging strategy revealed an association with the environmental gradient defined by RLQ axis 1.

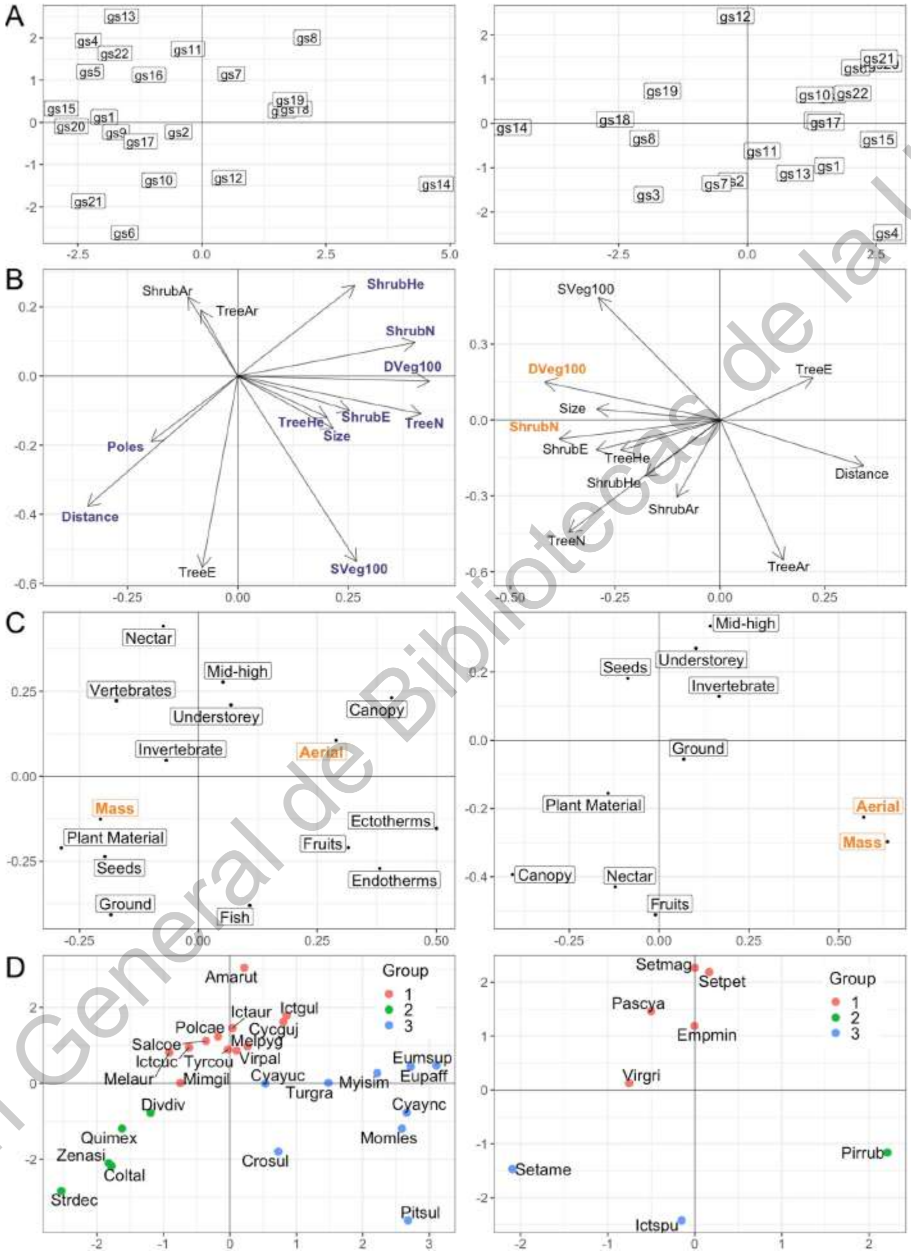

Figure 3. RLQ Analysis Results for Resident and Wintering Bird Species (Left and Right Columns, Respectively). Plots show the ordination of (A) surveyed sites, (B) environmental descriptors, (C) species' traits, and (D) species. Environmental descriptors and traits significantly associated to Axis 1 are shown with purple, while marginally significant associations are shown with orange labels. Bird species are grouped based on the output RLQ scores, using Ward's hierarchical clustering. Codes for environmental variables are shown in Table 1.

Classification of Species

Differences between subsequent values of Calinksi-Harabasz criterion suggested the clustering of resident species in three groups. The ordination plot showed that group 2 and 3 clearly occupied opposite extremes of the environmental gradient extracted by axis 1. Group 1 was the most numerous and included small-bodied species. This group comprised species that feed predominantly on the understorey and midhigh vegetation strata with a diet consisting mainly of invertebrates. On the contrary, group 2 included large-bodied birds that feed on invertebrates and seeds. Species belonging to this group tend to forage on the ground and use to a much lesser extent other strata. The third group includes medium-sized species with a more diversified use of food items and foraging strata.

Regarding wintering birds, species were clustered in three groups. Species classified in groups 1 and 3 were small-bodied birds. While species in group 1 were mainly insectivores and used the understorey and midhigh strata to forage, species belonging to group 3 were characterized by the use of the ground, understorey, and canopy strata to forage. Group 2 comprised Summer Tanager (*Piranga rubra*) only, a large-bodied species with a wider use of foraging strata and a predominantly insectivore diet.

Discussion

This study provided evidence that the functional composition of bird assemblages in urban green spaces can be linked to the characteristics of the sites. More precisely, our study revealed that trait composition of resident bird communities was influenced by several characteristics of green spaces during fall migration. This finding reinforces the well-known idea about the differing sensitivity of bird species to human-induced alterations related to the possession of particular traits (Sacco, Rui, Bergmann, Mueller, & Hartz, 2015). In the case of wintering species, our results do not support a significant relationship among the characteristics of green spaces and those species' traits that we assessed.

The ordinations obtained for resident and wintering species point to the existence of a common environmental gradient along which resident and wintering species distribute. This gradient was defined mainly by the isolation of green spaces but it was also related to the richness of plant species. In the case of resident birds, this was the dominant gradient since it extracted most of the co-variance. Isolation effects have been described for bird species richness in fragmented habitats including urban areas (Charre *et al*., 2013; MacGregor-Fors, Morales-Pérez, & Schondube, 2010; Martensen, Ribeiro, Banks-Leite, Prado & Metzger, 2012). Moreover, based on previous findings in forest landscapes and riparian forest parks we propose that species-dependent responses to isolation in urban green spaces are likely, and these responses could be determined by traits combinations (Atchinson & Rodewald, 2006; Martensen *et al*., 2012).

Our findings suggest that the richness of native trees and shrubs could be a relevant factor for wintering birds. Indeed, the effect of plant richness on taxonomic bird diversity has been well documented for urban areas including green spaces (Nielsen *et al*., 2014). With reference to the relevant role of plant species for urban birds, here, we contributed with evidence that native trees and shrubs seem to be the component of plant richness associated with functional composition of resident birds during the non-breeding season. This finding may be attributed to a larger complexity of the habitat in sites with a more diverse native flora, a favorable habitat condition for those species that require a wider diversity of resources such as food, shelter, and perches.

It is important to mention that exterior green spaces in the city of Merida tend to resemble more the original vegetation while inner green spaces usually consist of more landscaped sites. This urban landscape pattern may have important conservation implications since it may be indirectly driving the distribution of species based on their functional traits. There is evidence of the influence of the location of green spaces on its characteristics. For example, those green spaces located in the more urbanized regions of a city tend to be smaller and to include more exotic plant species (Useni-Sikuzani *et al*., 2018). Hence, we considered that the negative relationship between plant species richness and isolation of green spaces merits deeper examination. Though the inclusion of green spaces of different type may imply confounding factors, we believe our findings are valuable because they suggest the existence of an interplay of factors still scarcely understood.

With regard to vegetation cover within green spaces, the models selected for both resident and wintering birds did not support its contribution to the functional composition of the communities. This was an unexpected result given the existing evidence of vegetation cover as a determinant of several dimensions of bird diversity (Cristaldi, Giraudo, Arzamendia, Bellini, & Claus, 2017; Harvey *et al*., 2006). We suggest that further research is needed to investigate the relative contribution of vegetation cover, especially, shrub and tree cover to bird trait composition within the Neotropical realm, especially in tropical deciduous forests.

Contrary to our expectations, we did not identify clear trait syndromes related to environmental gradient but our results showed that there was a shift in body mass at the community level along the more-to-less isolated gradient. Results suggest that resident species were filtered along this gradient based on their body size and the strata in which they forage: as green spaces became more isolated smaller birds became uncommon.

When studying the consequences of urban-driven habitat transformation on biodiversity, trait-based approaches allow to obtain more generalizable conclusions by using a set of traits, rather than organisms' taxonomic identity (Dray & Legendre, 2008; McGill, Enquist, Weiher, & Westoby, 2006). In accordance with this conceptual approach, species' trait levels that are subject to the filtering of the environment have been identified for urban birds (Lees & Moura, 2017; Lim & Sodhi, 2004). For instance, omnivorous and insectivorous species were the most frequently encountered species in urban parks of Porto Alegre, Brazil (Scherer *et al*., 2015). While works that employ a descriptive approach are relatively common, studies that statistically test the link between traits and environment for urban bird communities are scarce (but see Sacco *et al*., 2015). Currently, RLQ and fourth-corner tests represent an integrated approach to analyze trait-environment relationships and to determine functional groups (Kleyer *et al*., 2012). We highlight the fact that RLQ and fourth-corner tests assess trait-environmental relationships considering either trait syndromes or environmental gradients, and not just single traits or environmental descriptors (Almeida *et al*., 2018; Gamez-Virues *et al*., 2015). Results obtained this way may deepen our understanding of the mechanisms underlying the sorting of species in the urban environment. So, future research on the assembly of urban bird communities should combine taxonomic and trait information so ecological knowledge advances towards a more general and predictive one (Webb, Hoeting, Ames, Pyne, & Poff, 2010).

Implications for Conservation

To our knowledge, this is the first published study about urban avifauna in YP. Considering urbanization trends, we believe that there is an urgent need to investigate the effects of urbanization in Mexico, especially in areas of evergreen and deciduous tropical forests, the vegetation types with the largest percentages of species richness in Mexico (Navarro-Sigüenza *et al*., 2014) and that differ from the temperate forest for which information is more abundant. Near one fifth (19.0 %) of the estimated species richness for Yucatan was recorded in this study (Navarro-Sigüenza *et al*., 2014) together with 21.4 % of the endemic species of YP. Lees and Moura (2017) registered a similar percentage of the regional species pool for the city of Belém, in the Brazilian Amazon. We highlight the relevance of this in the context of migratory species that rest and feed in the area either during the whole winter or during a brief period (Deppe & Rotenberry, 2005). Besides, we emphasize the contribution of waterbodies to bird diversity of the city. Though these waterbodies do not occur naturally, they provide habitat for both resident and migratory species, and this deserves attention considering the karstic origin of the Yucatan Peninsula and the scarcity of waterbodies in the area (Torrescano-Valle & Folan, 2015).

We strongly recommend to maintain urban green spaces of varying habitat characteristics that comprise from remnants of the original vegetation to landscaped sites. Our results show that habitat characteristics within green spaces and in their vicinity, can affect functional composition of bird assemblages. We caution that severe alterations of the habitat can reduce the abundance of species that possess particular traits and this can affect ecosystem functioning (Bovo *et al*., 2018). Finally, we want to invite urban planners to acknowledge: 1) the value that green spaces hold for biodiversity (Carbó-Ramírez & Zuria, 2011) besides its function as public spaces intended for people use, 2) the fact that urban green spaces may represent the only opportunity of locals to experience close contact with wildlife, and 3) that ecological knowledge should be applied to enhance biodiversity in green spaces. If green spaces are intended to conserve local biodiversity local authorities need to issue guidelines and to set up mechanisms aimed to regulate the management of urban green spaces.

CAPÍTULO 3. Taxonomic, functional and phylogenetic diversity of bird assemblages in urban green spaces: temporal variation and their ecological drivers.

Introduction

Habitat loss and modification resulting from human activity are among the dominant drivers of the current mass extinction of fauna, the so-called Anthropocene defaunation (Dirzo *et al*., 2014; Young *et al.*, 2016). Considered a long-lasting transformation, urban growth contributes to the loss of natural habitats (MacDonald *et al.*, 2020) and leads to declines in local species richness and abundance (Newbold *et al.*, 2015). For the last decades, urban growth rates tended to be higher than urban population growth rates (Seto *et al.*, 2011). Given the global expansion of urban areas, we require a better understanding of those factors that support the conservation of biodiversity in these areas.

Much of the urban impacts on biodiversity research has focused on birds (Magle *et al.*, 2012; Beninde *et al.*, 2015). Studies have mainly reported reductions in species richness and species composition changes (Carvajal-Castro *et al.*, 2009; Di Pietro *et al*., 2021 but see Benitez *et al*., 2021). Species richness has been a widely-used measure of biodiversity to assess the effect of urbanization on bird communities (Rush *et al*., 2014; Canedoli *et al*., 2017; Matthies *et al*., 2017). However, the assumption of species being equivalent may not be ecologically precise (Chave, 2004). In addition, the multidimensional nature of biodiversity (Naeem *et a*l., 2016) has been acknowledged in urban bird research in recent years, and studies have increasingly included measures of functional, genetic, and phylogenetic diversity (Filippi-Codaccioni *et al*., 2009; Delaney *et al*., 2010; Morelli *et al*., 2016; Schmidt *et al*., 2020).

In this study, we use taxonomic, phylogenetic, and functional measures as indicators of diversity to get a more complete insight into the response of bird diversity to urban habitats. Functional diversity can be defined as the diversity of functional traits of those species found in an assemblage (Mason and Mouillot, 2013). A functional trait is any phenotypic characteristic that can influence ecosystem processes or the species' response to the environmental conditions (Hooper *et al*., 2005). Phylogenetic diversity arose also as an alternative to quantify the biological attributes in an assemblage (Faith, 1992) but it differs from functional diversity in the fact that it comprises observed and unobserved characters (Faith, 2002). To do so, the diversity of features is estimated from the phylogenetic relationships among the species under the assumption that estimated amounts of divergence, measured as the branching pattern and branch lengths in a cladogram, can be used to derive feature diversity (Faith, 1994).

Apart from its inherent value, functional and phylogenetic diversity hold conceptual relevance for community ecology since functional and phylogenetic clustering patterns can provide insights into community assembly mechanisms such as the environmental filtering of species from the regional species pool (Weiher, 2011). In this regard, documented reductions in phylogenetic and functional diversity in urban bird assemblages (Sol *et al.*, 2017; La Sorte *et al.*, 2018; Leveau and Leveau, 2020) may imply the filtering of species but it is necessary to test if diminished values are different from random expectations or are due only to reductions in species numbers (Swenson, 2014).

Previous works conducted in human-modified areas show that the environment and landscape effects on biological diversity can be dimension specific (Cisneros *et al.*, 2015). In agreement with this, it has been reported that urbanization can lead to decoupled changes of bird diversity dimensions when these are simultaneously assessed (Morelli *et al.*, 2017; Morelli *et al.*, 2018; Ibáñez-Álamo *et al.*, 2020). Furthermore, bird diversity dimensions can vary among seasons in urban areas (Alexander *et al*., 2019; Ibáñez-Álamo *et al.*, 2020). Therefore, to achieve an effective urban bird conservation, it is necessary to employ a multidimensional approach with a temporal perspective for its diversity assessment (Devictor *et al.*, 2010).

Regarding the factors that can influence urban bird diversity, it has been proposed that the assembly of urban local biotas may result from hierarchical processes occurring at different scales (Aronson *et al.*, 2016). Previous works support this notion and have shown that a wide range of factors operating at different spatial scales can influence bird diversity in urban green spaces (Yang *et al.*, 2015; Morelli *et al.*, 2018; Liu *et al.*, 2019). In this context, we aimed to examine the taxonomic, phylogenetic, and functional bird diversity together with its determinants. We conducted the study in urban green spaces, because they represent available habitat for a wide range of organisms, including birds (Cornelis and Hermy, 2004;

MacGregor-Fors *et al.*, 2016). This study considered two spatial scales, the sampling unit and the green space (hereafter point level and site level, respectively). These scales correspond to the finest resolution levels at which diversity can be described (Magurran, 2004). Both scales were assessed because we consider relevant to examine the diversity patterns and their relation to environmental determinants separately at each scale. On the one hand, we described bird diversity at the site scale and explored the influence of factors associated with its size, form, isolation and habitat heterogeneity. On the other hand, we evaluated bird diversity at the point scale and study the following factors influence: vegetation, ground cover types, urban equipment, and anthropic disturbance.

Based on the abovementioned information, we expected bird diversity dimensions to vary among seasons. Under the assumption that urbanization acts as an ecological filter for species and traits, we expected to detect a departure from randomness in phylogenetic and functional diversity observed values. More specifically, we hypothesized that bird assemblages would show a clustered phylogenetic and functional structure. We also hypothesized that bird diversity, measured in any of its dimensions, would decrease as the degree of transformation of green spaces increased, and that this pattern could be detected at the site and point scale. Here, we did not compute a transformation index but we measured variables that can be used as surrogates for transformation degree at the site and point scales, such as distance from wild areas and urban equipment respectively. Regarding biodiversity-environment relationships at the site scale, we expected habitat heterogeneity to positively affect taxonomic, phylogenetic, and functional diversity based on existing information (Benitez *et al.*, 2021). A last hypothesis concerned seasonal variation: studied bird diversity dimensions and their relationship with environmental factors would differ among seasons.

Methods

Study site

We conducted this study in Mexico City, formerly called Distrito Federal, which forms part of the Metropolitan Area of the Valley of Mexico. Mexico City has an approximate area of 1,499 km^2 and an estimated population of 9,209,944 inhabitants (INEGI, 2021) and despite its high urbanization degree, near one third of its extent is covered by natural vegetation (471 km^2) (INEGI, 2016). The climate of Mexico City varies geographically, being template towards the northeastern part of the city and semi cold towards the southwestern part. Hence, mean annual temperate ranges between 8 and 10ºC in the southwestern region and can be over 16ºC in the northeastern part (Hernández-Cerda *et al*., 2016). The dry season occurs from October to May and the rainy season from June to September (INEGI, 2017). This study was performed in Mexico City for the following reasons: (i) green spaces with heterogeneous environmental conditions are distributed throughout the city and (ii) the matrix that surrounds green spaces varies from mostly-impervious to mostly-vegetated surfaces which permitted us to assess the influence of the surrounding landscape.

We surveyed 32 green spaces distributed from near downtown to the southwestern urban fringe (Fig. 1). Surveyed green spaces included recreational parks, an arboretum, a university campus and natural protected areas. Site selection was intended to encompass variation of environmental factors along a transformation gradient. At one extreme, plant dominant species were conifers that belonged to the genus *Pinus* ("pine") and *Abies* ("fir") and artificial structures such as electric poles were scarce or absent. At the other extreme, the vegetation was dominated by the broadleaf trees *Fraxinus uhdei* ("ash") and *Jacaranda mimosifolia* ("blue jacaranda") and the coniferous tree *Cupressus lusitanica* ("cypress"). In these sites, artificial structures were abundant and natural ground covers were substituted by concrete to differing extents.

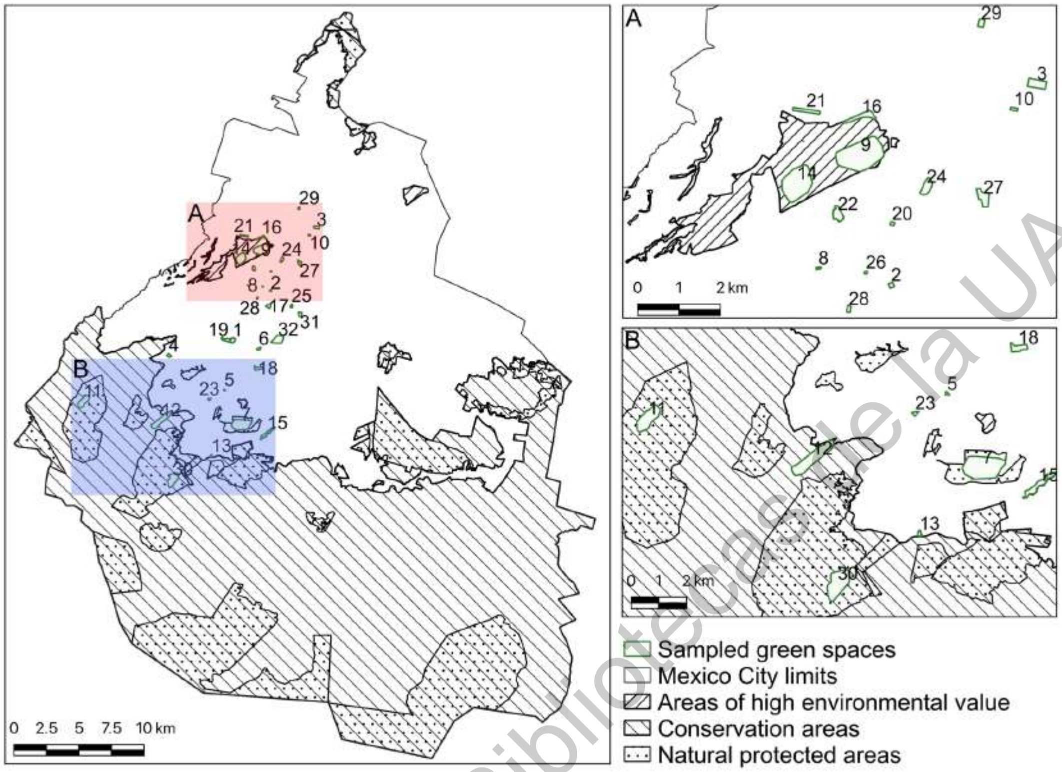

Fig. 1. Distribution of sampled green spaces and areas considered as potential species pools in Mexico City.

Bird surveys

Birds were counted by a single observer (RND) in three different periods of 2018 which corresponded to: (1) the wintering season (from January 3rd to January 22nd), (2) the dry part of the breeding season (from May 1st to May 27th) and the rainy part of the breeding season (from August 1st to September 4th). Hereafter, we will refer to these periods as winter, dry breeding, and wet breeding seasons. Data on bird richness and abundance were collected within four hours after sunrise and by using standardized 5-minute point counts with a fixed radius of 25 m. Points were separated by at least 200 meters to avoid double-counting of individual birds (Bibby *et al.*, 1993). All points were visited twice on different days in a random order. Point counts were used to obtain a reliable species' relative abundance estimate (Bibby *et al.*, 1993) and for each point count and each species, the highest number of individuals recorded in any of the two visits was used as the abundance estimate. The number of points located in each green space ranged from one to nine based on green space size and accessibility. In total, we sampled 110 points distributed in thirty-two green spaces. We recorded all birds detected visually or acoustically, except those that were flying over.

Site-level explanatory variables

To determine the size of green spaces we delineated their perimeters with the aid of a Geographic Information System (QGIS) and high resolution images (Google Earth). Shape index was calculated with the following formula (Corey *et al.*, 2012):

$$Index = \frac{perimeter}{2\sqrt{(\pi \times area)}}$$

Shape index ranges from 1 to ∞. A shape index of 1 corresponds to a circular patch, whereas an irregular patch has a shape index > 1. As the shape of the patch becomes more irregular, the shape index increases (Corey *et al.*, 2012). To describe the distribution and amount of vegetation in each green space we used satellite imagery to derive a vegetation index. We computed the Normalized Vegetation Index (NDVI), an index that relies on the reflectance of red and near-infrared light (Huete *et al.*, 1985). NDVI has been used for different purposes, including vegetation cover and vegetation density estimation (MacDonald *et al.*, 2010; Odindi and Mhangara, 2012). In this work, NDVI was derived from Sentinel-2A multi-spectral images obtained from the European Space Agency. In order to reduce discrepancies, we used images whose acquisition date was as close as possible to the bird sampling seasons while maintaining a low cloud cover percentage. For winter, dry breeding, and wet breeding seasons the acquisition dates of images were January 7th, June 1st and July 26th 2018 respectively. NDVI was calculated from the Red and Near-infrared (NIR) bands which have a spatial resolution of 10 m using the following formula:

$$NDVI = \frac{NIR - RED}{NIR + RED}$$

NDVI is dimensionless and ranges from -1 to +1. Negative values indicate the absence of vegetation while positive values represent vegetated areas. NDVI values can be categorized into useful classes that separate different conditions of vegetation. Values above 0.2 are associated with vegetated areas and the larger the NDVI values the denser the vegetation (Fung and Siu, 2001; Hashim *et al*., 2019). We followed Odindi and Mhangara (2012) to group the NDVI values into five classes that describe different vegetation conditions: unvegetated ($\text{NDVI} \leq 0$), very sparse vegetation ($0 > \text{NDVI} < 0.2$), sparse vegetation ($0.2 \geq \text{NDVI} < 0.4$), dense vegetation ($0.4 \geq \text{NDVI} < 0.6$), and very dense vegetation ($0.6 \geq \text{NDVI} \leq 1$).

For every studied season, two variables were derived from the corresponding NDVI data to describe vegetation cover of each green space. The first variable was the densely-vegetated fraction (DVF), estimated as a function of the number of pixels that represent dense and very dense vegetation. NDVI values above 0.5 are considered as fully vegetated pixels (Sobrino *et al.*, 2004), hence, DVF is a close approximation of the proportion of fully vegetated area in the sampled polygon. Densely-vegetated fraction was computed as follows (Wong *et al.*, 2019):

$$\text{DVF} = \frac{\text{Count of Dense vegetation pixels} + \text{Count of Very dense vegetation pixels}}{\text{Total count of pixels in the sampled polygon}}$$

The second variable based on NDVI was the diversity of classes describing vegetation condition and this was used as a surrogate of habitat diversity (HD). Simpson's diversity index was used to quantify the diversity of vegetation conditions (Nagendra, 2002). The index was calculated with the following formula:

$$Simpson's\ diversity\ index = 1 - \sum_{i=1}^{N} {p_i}^2$$

where p_i is the proportional abundance of the *i*th NDVI class.

Additionally, we calculated the DVF for 50-m, 100-m and 200-m width buffers around each green space. These estimations did not include estimations of DVF within the green space, so the information of each buffer is spatially exclusive from that of the green space (the "doughnut" method according to Dunford and Freemark, 2005). We used the aforementioned buffers because landscape variables measured to this extent have been reported as significant drivers of bird communities (Carbó-Ramírez and Zuria, 2011) and because we considered that management interventions at these small scales are more likely to be accomplished in urban landscapes. We used the distance between each green space and its closest likely source of species as a proxy of isolation (Fernandez-Juricic, 2000). We assumed protected areas, environmental valuable areas, and areas designated for conservation to function as regional species pools (Fig. 1). We measured the Euclidean distance from the centroid of each green space to the edge of the closest species pool as long as it was larger than 100 ha.

Point-level explanatory variables

A number of variables related to the physical environment and disturbance regime were measured at each sampling point to characterize the habitat. These variables were selected to describe four relevant features of urban habitats for birds (Evans *et al.* 2009; MacGregor-Fors and Schondube, 2011): vegetation, ground cover types, urban infrastructure, and anthropic disturbance. Information on vegetation included tree species richness, diameter of trees at breast, and height of trees. Diameter and height of trees were measured for a maximum of thirteen randomly selected trees. Additionally, we visually estimated the percent extent cover of bare ground, grass, shrubs and trees. Cover estimation was performed in the field because remote data fail to capture information about overlapped vegetation strata or other objects (Leveau *et al.*, 2018). To describe the extent to which places were equipped with infrastructure for human use, we recorded the number of the following: litter bins, benches, electric wires, electric poles, lamps on poles, and lamps on the ground. To characterize anthropic disturbance, we counted the number of pedestrians and dogs encountered during each point count. We also registered instantaneous noise levels using a sound level meter (AR814 Intell Instruments Plus) every ten seconds during one minute, before and after each bird count.

All urban infrastructure descriptors were combined into a composite variable employing the simple averaging approach (Song *et al.* 2013). In this approach, the composite variable is obtained by summing *z* scores of the original variables. Previous analyses, (not shown here) indicated that this procedure yielded results similar to those obtained with principal component analysis. For noise characterization, we followed Gonzalez-Oreja *et al.* (2012) and estimated the frequency distribution of noise values. For every sampling unit, we pooled data and through a resampling procedure, the equivalent noise levels (LEQ) for 1 min were computed. The resampling procedure was repeated 1000 times to estimate the frequency distribution of LEQ. The 10th percentile of the frequency distribution (L_{90}) was used in the statistical analysis. LEQ is considered an appropriate measure for steady environmental noise (Patter *et al*., 2009).

Trait data

We employed four traits to characterize all detected species: log-transformed body mass, main diet item, foraging strategy, and foraging stratum. Main diet items were the following: aquatic animals, fruit, generalist, invertebrates, nectar, seeds, and vertebrates. The foraging strategy categories were: aerial sallying, aerial screening, arboreal gleaning, bark gleaning, foraging generalist, and ground foraging. We considered the following foraging stratum categories: aerial, canopy, ground, midhigh, understory, around water surface, below water surface. Foraging stratum is expressed as the estimated percentage use of each one of the all strata considered. The values of all the strata sum to 100. We selected these traits because their occurrence can be influenced by environmental conditions (Castaño-Villa *et al.*, 2019). We built a species-trait matrix with information sourced from previous publications (Wilman *et al.*, 2014; Tobias and Pigot, 2019).

Community diversity metrics

For each season, we quantified diversity with three different metrics for both green spaces and sampling points. Diversity metrics at the sampling point level were based on cumulative species lists for the species detected on both visits. In the case of green spaces, diversity metrics were calculated after the species lists of all the corresponding sampling points were pooled. We calculated taxonomic diversity as species richness. To quantify functional diversity we used functional richness, which measures the amount of trait space occupied by the species in the community (Villéger *et al.*, 2008). For functional richness to be computed, the number of species recorded in assemblages must be at least three. Phylogenetic diversity was quantified by means of the Faith's phylogenetic diversity which measures the total evolutionary history of the species in a community (Frishkoff *et al.*, 2014). Faith's phylogenetic diversity is calculated as the sum of all branch lengths spanned by the tree connecting all the species in a community (Faith, 1992). For phylogenetic metric calculation, we used 1,000 dated phylogenies (Jetz *et al.*, 2012): 500 phylogenies using the Hackett backbone and 500 phylogenies using the Erikson backbone following La Sorte *et al.* (2018). The use of sets of phylogenies can provide more robust comparative inferences than the use of a single consensus tree (Rubolini *et al*., 2015). Once computed, we averaged the values across the 2000 phylogenies.

Data analysis

Faith's phylogenetic diversity and functional richness tend to be positively correlated with species richness (Swenson, 2014; Voskamp *et al.*, 2017). To remove the effects of species richness on both indexes, we generated 999 null assemblages. Null assemblages were obtained through the randomization of species names on the trait data matrix and on the phylogeny maintaining constant the number of species of each focal assemblage (Swenson, 2014). For each season, we used a phylogeny and a trait matrix that included the species recorded in the specific season. To compare observed Faith's phylogenetic diversity and functional richness values with expected values from null models we calculated standardized effect sizes (SES) and these were obtained using the following formula:

$$\text{SES}=\frac{\text{Mean}_{\text{obs}}-\text{Mean}_{\text{rand}}}{\text{SD}_{\text{rand}}}$$

where Mean_{obs} is the observed diversity value, $\text{Mean}_{\text{rand}}$ is the mean value of the randomized assemblages, and SD_{rand} is the standard deviation of the randomized assemblages. Negative SES values indicate an observed value lower than the average value expected by chance, while positive values evidence an observed value higher than the average value expected by chance (Swenson, 2014). In terms of community structure, negative values suggest phylogenetic clustering or trait convergence, while positive SES values suggest phylogenetic over-dispersion or trait divergence. In this way, SES values provide additional information to that obtained from non-standardized diversity measures. To test if observed values were either significantly lower or higher than the average value expected by chance, we assumed a normal distribution of observed and random values together. For a significance level = 0.05, observed values are considered significantly if they are less than or equal to -1.96, or greater than or equal to 1.96 (Zar, 2014).

To assess the relationships between the response variables and the explanatory variables (Table 1), we used an information theoretic approach. This approach ranks and weights candidate statistical models that function as biological hypotheses (Grueber *et al.*, 2011). Previous to model fitting, we followed standard procedures in multiple regression analysis. We explored the distribution of the response variables to select the appropriate error structure. Explanatory variables were standardized to mean = 0 and standard deviation = 1. We checked for potential multicollinearity between explanatory variables by means of the Variance Inflation Factor (VIF > 3.5) and the Spearman correlation (r > 0.70) and we removed one variable of each strongly correlated pair of variables. Models of species richness were fitted using a Poisson distribution, and models of functional richness, Faith's phylogenetic diversity, SES functional richness, and SES Faith's phylogenetic diversity employed a Gaussian distribution. For those models with a Poisson distribution, we checked for potential over-dispersion. In case it was detected, we corrected standard errors using a quasi-GLM model (Zuur *et al.*, 2009).

Table 1. Predictors included in models to describe variation in response variables.

Code	Variable (Source)
Site-level descriptors	
Area	Area of the green space (GIS)
DVF_0	Densely-vegetated fraction within the green space (GIS)
Habitat Diversity	Habitat diversity within the green space (GIS)
Isolation	Euclidean distance from the closest species pool (GIS)
Tree richness	Tree species richness within the green space (Field data)
Shape	Shape index of the green space (GIS)
DVF_{50}	Densely-vegetated fraction in a 50-meter buffer around the green space without including DVF_0 estimations (GIS)
DVF_{100}	Densely-vegetated fraction in a 100-meter buffer around the green space without including DVF_0 estimations (GIS)
DVF_{200}	Densely-vegetated fraction in a 200-meter buffer around the green space without including DVF_0 estimations (GIS)
Point-level descriptors	
Tree richness	Plant species richness in the sampling point (Field data)
Diameter at Breast	Coefficient of variation of diameter at breast of trees in the sampling point (Field data)
Tree Height	Standard deviation of height of trees in the sampling point (Field data)
Bare ground	Percent extent cover of bare ground (Field data)
Grass	Percent extent cover of grass (Field data)
Shrub	Percent extent cover of shrubs (Field data)
Tree	Percent extent cover of trees (Field data)
Equipment	Equipment found in sampling points expressed as a composite variable including the number litter bins, benches, electric wires, electric poles, lamps on poles, and lamps on the ground (Field data)
People	Maximum number of visitors registered in either of the two visits (Field data)
Dog	Maximum number of dogs registered in either of the two visits (Field data)
Noise	Noise levels expressed as the 10th percentile of the frequency distribution of noise values obtained through a resampling procedure (Field data)

At the site level, we fitted generalized linear models (GLM). For each diversity metric x season combination (for example, functional richness – wet breeding season) we fitted a global model including all the variables measured at the green space (*i.e.* isolation, tree richness, shape index, densely-vegetated fraction, and habitat diversity). Three additional models were fitted, each one incorporating the densely-vegetated fraction for one of the abovementioned buffers. In this way, four global models were fitted for every metric x season combination.

We selected the most parsimonious model using the Akaike information criterion modified for small sample size (AIC_c) (Symonds and Moussalli, 2011). From the set of the four candidate global models, we selected the top-ranked model, that is, the model with the lowest AIC_c value. To quantify the relative performance of the models, we calculated the difference (Δ_i) between the best rated model and the other models. Additionally, based on Δ_i we computed two measures to quantify the relative support for each model. The first one, the Akaike weight (w_i) measures the weight of evidence that supports model *i* as being the best model, given the data and the set of candidate models (Burnham and Anderson, 2002). Some authors interpret the Akaike weight as the probability that model *i* is indeed the best model. The second measure, the Evidence Ratio compares the likelihood of the top-ranked model with the likelihood of model *i* (Symonds and Moussalli, 2011).

Once a global model was selected, we fitted all possible submodels following an all-subset approach. The full set of submodels equals 2^n, where n is the number of parameters. Then, we performed model selection including the 95% confidence set of models. To account for model uncertainty, with the selected set, we implemented model averaging using the two existing methods to perform model averaging (Symonds and Moussalli, 2011). Natural averaging, averages the estimator β_i just over the models where the predictor *i* appears. This method is recommended when there is strong support for a model ($w_i > 0.9$). The other method, zero averaging, averages β_i over the full set of models. In those models where the predictor *i* is absent, a value = 0 is used in the calculation of the average. This makes β_i shrink towards 0 for those predictors that appear in the less supported models (Burnham and Anderson, 2002; Grueber *et al.*, 2011)

At the sampling point level, variation in biodiversity indices was modelled by means of generalized linear mixed models (GLMM) given the nested structure of the data. Habitat variables were set as fixed effects while green space was the random blocking factor. Model ranking and model averaging were performed as described previously. This procedure was implemented for every response variable.

All statistical analyses were performed in R version 3.5.2 (R Core Team, 2018). To obtain phylogenetic and functional diversity we employed functions found in the libraries “picante” (Kembel *et al.*, 2010) and “FD” (Laliberté *et al.*, 2014). The generation of candidate subset models, model ranking and model averaging were implemented with functions found in the library “MuMIn” (Bartón, 2020). GLM were fitted using core functions found in R and GLMM were implemented using functions from the library “lme4” (Bates *et al.*, 2015).

Results

Diversity patterns

A total of 91 species from 28 families were recorded across all sites and seasons with a mean of 10 species per site (± 6) and a mean of 5 species per point (± 2.8). The number of detected species decreased from winter (71 species) to the dry breeding season (63 species), and then to the wet breeding season (55 species). While thirty-seven species were recorded across seasons, some species were recorded in just one season (Appendix 1). A phylogeny of all detected species can be seen in Fig. 2. The number of individuals was similar for winter and early-breeding seasons (1421 and 1437 individuals, respectively) but decreased for the late-breeding season (1021 individuals).

All the three diversity dimensions showed a similar trend across the year. On average, diversity values were higher for the winter season and decreased towards the early and the late breeding seasons (Table 2, Fig. 3).

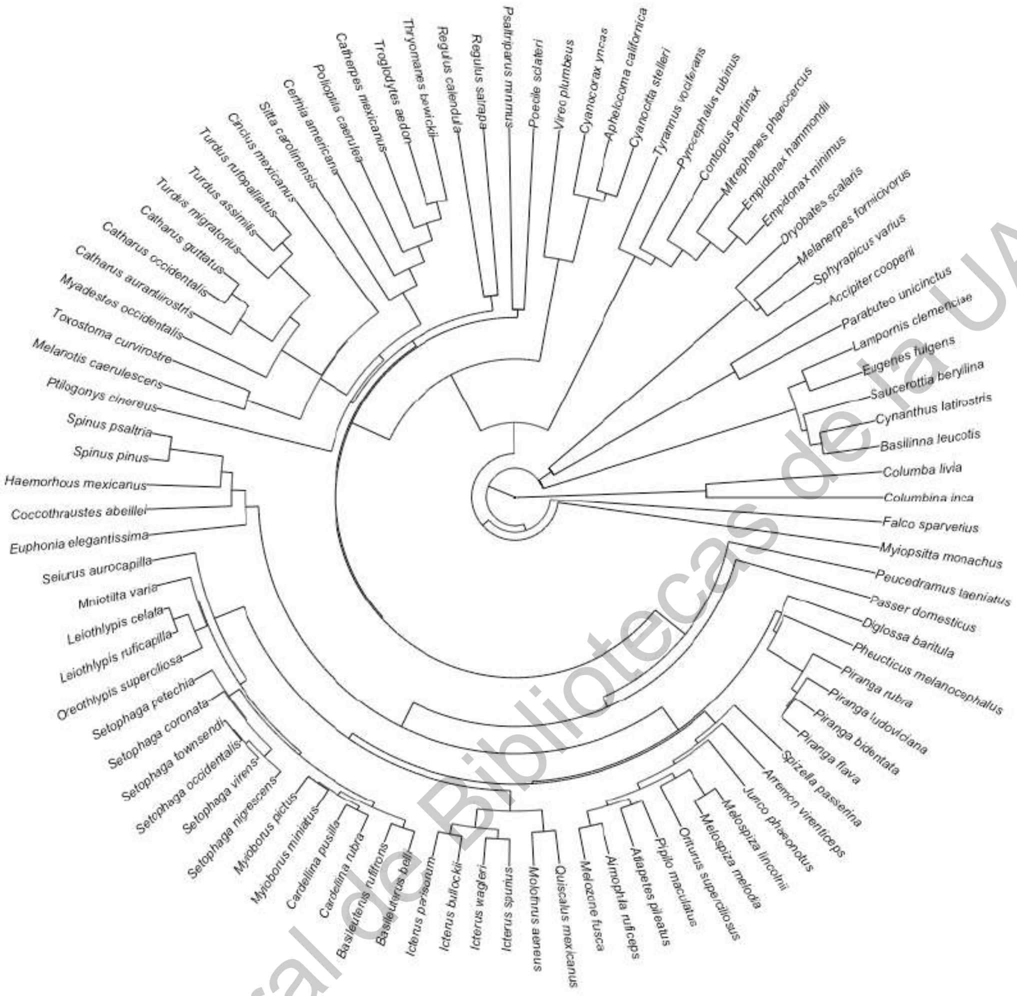

Fig. 2. Phylogeny of detected birds in sampled green areas. The displayed phylogeny is one of the 1000 phylogenies used to compute phylogenetic diversity (see Methods).

Table 2. Mean values ± standard deviation (minimum – maximum) of taxonomic, functional, and phylogenetic diversity of bird communities in green spaces in each of the studied seasons.

Diversity index	Winter	Dry breeding	Wet breeding
Species richness	12.3 ± 7.7 (3 - 41)	10.8 ± 4.9 (2 - 26)	8.4 ± 5.2 (3 – 27)
Faith's phylogenetic diversity	470.6 ± 192.5 (107.1 – 1008.6)	452.6 ± 142.0 (88.3 – 846.0)	385.0 ± 150.1 (117.6 – 772.6)
Functional richness	0.084 ± 0.045 (0.001 – 0.163)	0.010 ± 0.007 (0.00002 – 0.025)	0.073 ± 0.038 (0.006 – 0.145)

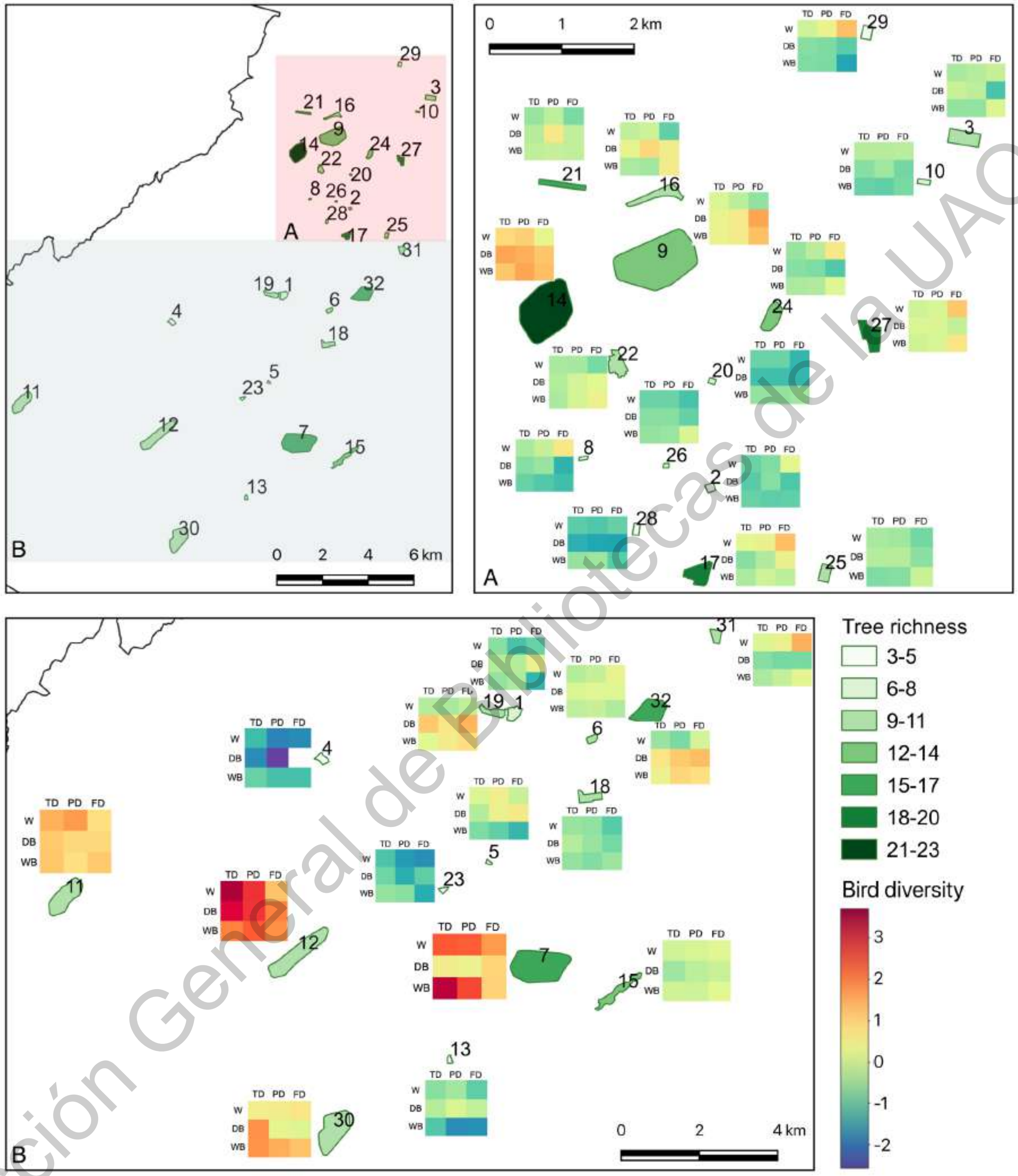

Fig. 3. Location of sampled green spaces in Mexico City. Tree richness is represented by the color of the polygon. Taxonomic diversity (TD), phylogenetic diversity (PD) and functional diversity (FD) of each green space during winter (W), dry breeding (DB) and wet breeding seasons (WB) are represented as a heatmap. Diversity values were standardized for each diversity dimension-season combination. A white square is used when functional diversity could not be computed.

For all studied seasons, taxonomic diversity was significantly correlated with both phylogenetic and functional diversity (Fig. 4 and 5). The correlation strength with the taxonomic diversity was moderate for the functional diversity and strong for the phylogenetic diversity. In contrast, standardized effect sizes of functional and phylogenetic diversity were not correlated with taxonomic diversity at the site level. When considering sampling-point correlations, SES values of both phylogenetic and functional diversity showed a weak but significant correlation with taxonomic diversity.

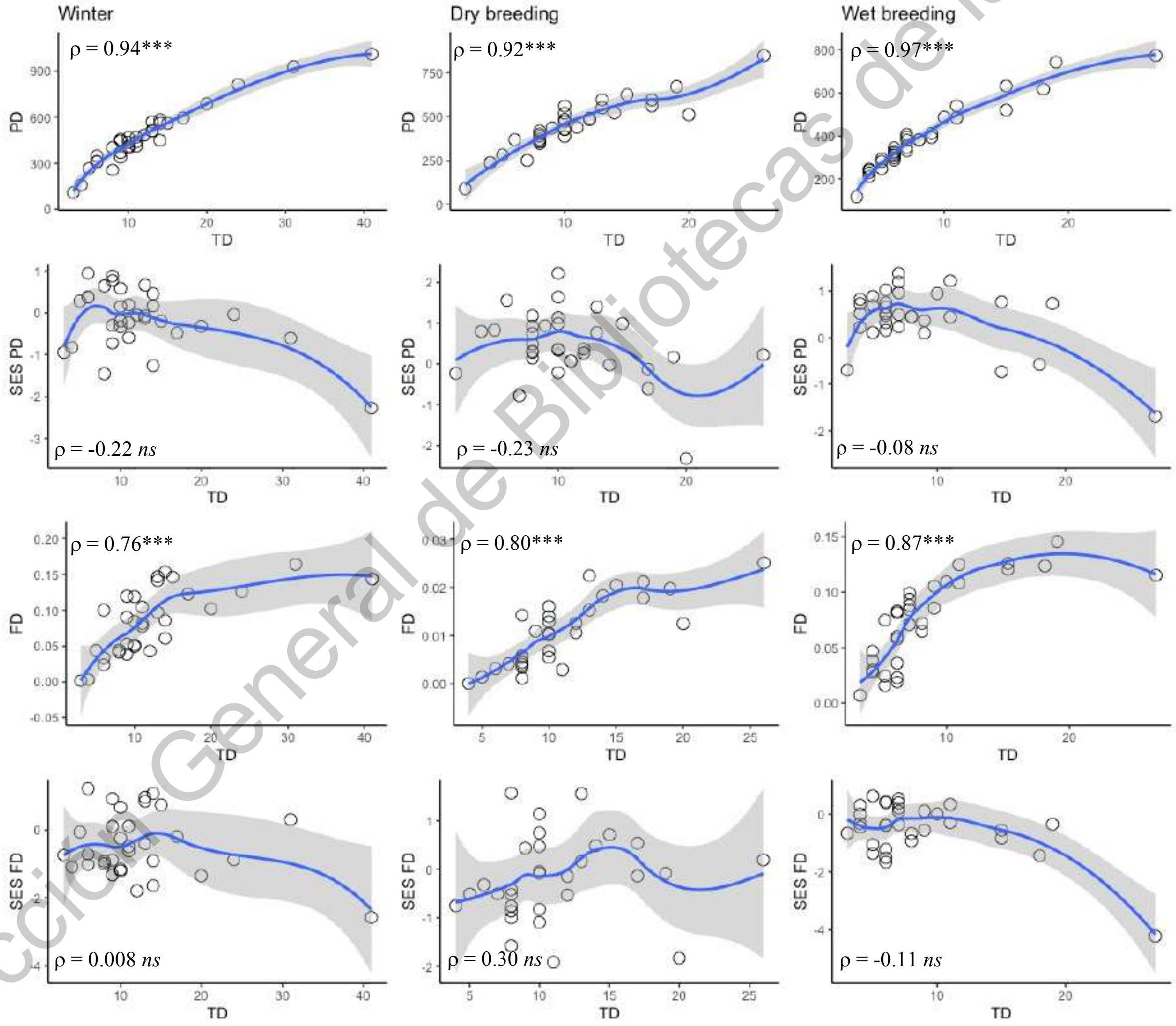

Fig. 4. Relationship between measures of taxonomic (TD), phylogenetic (PD), and functional diversity (FD) for bird communities in green spaces during winter, dry breeding and wet breading seasons at the site level. Strength of the relationship is expressed as Spearmans's correlation coefficient ρ (*ns P* ≥ 0.05, **P* < 0.05, ***P* < 0.01, ****P* < 0.001).

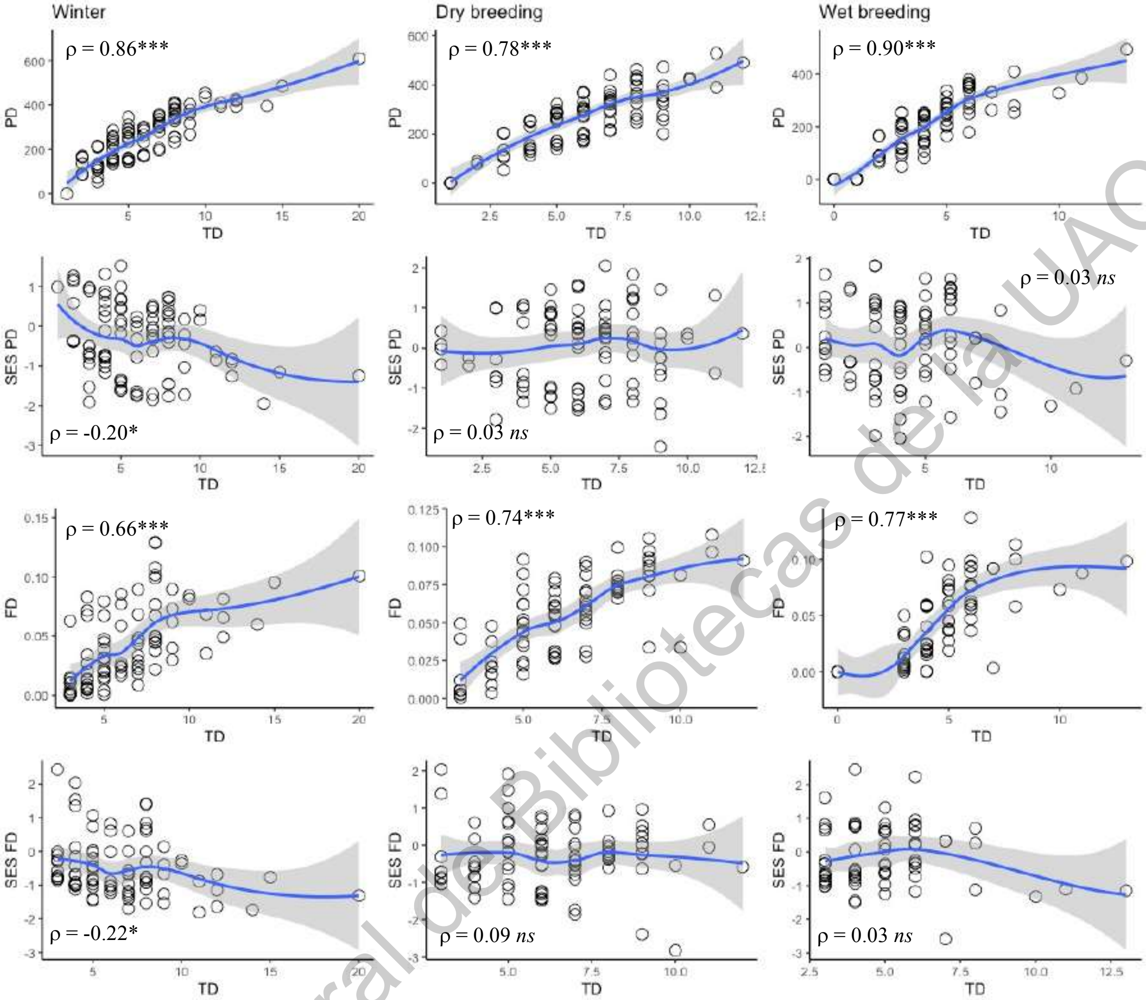

Fig. 5. Relationship between measures of measures of taxonomic (TD), phylogenetic (PD), and functional diversity (FD) for bird communities at the sampling-point level during winter, dry breeding and wet breading seasons at the sampling point level. Strength of the relationship is expressed as Spearmans's correlation coefficient ρ (*ns P* ≥ 0.05, **P* < 0.05, ***P* < 0.01, ****P* < 0.001).

Overall, functional and phylogenetic structure of bird communities in green spaces were neither clustered (SES ≤ -1.96) nor over-dispersed (SES ≥ 1.96) in any of the studied seasons. For both functional and phylogenetic diversity, less than 5% of the communities showed a structure pattern different from that expected by chance based on their species richness (Figs. 6 and 7).

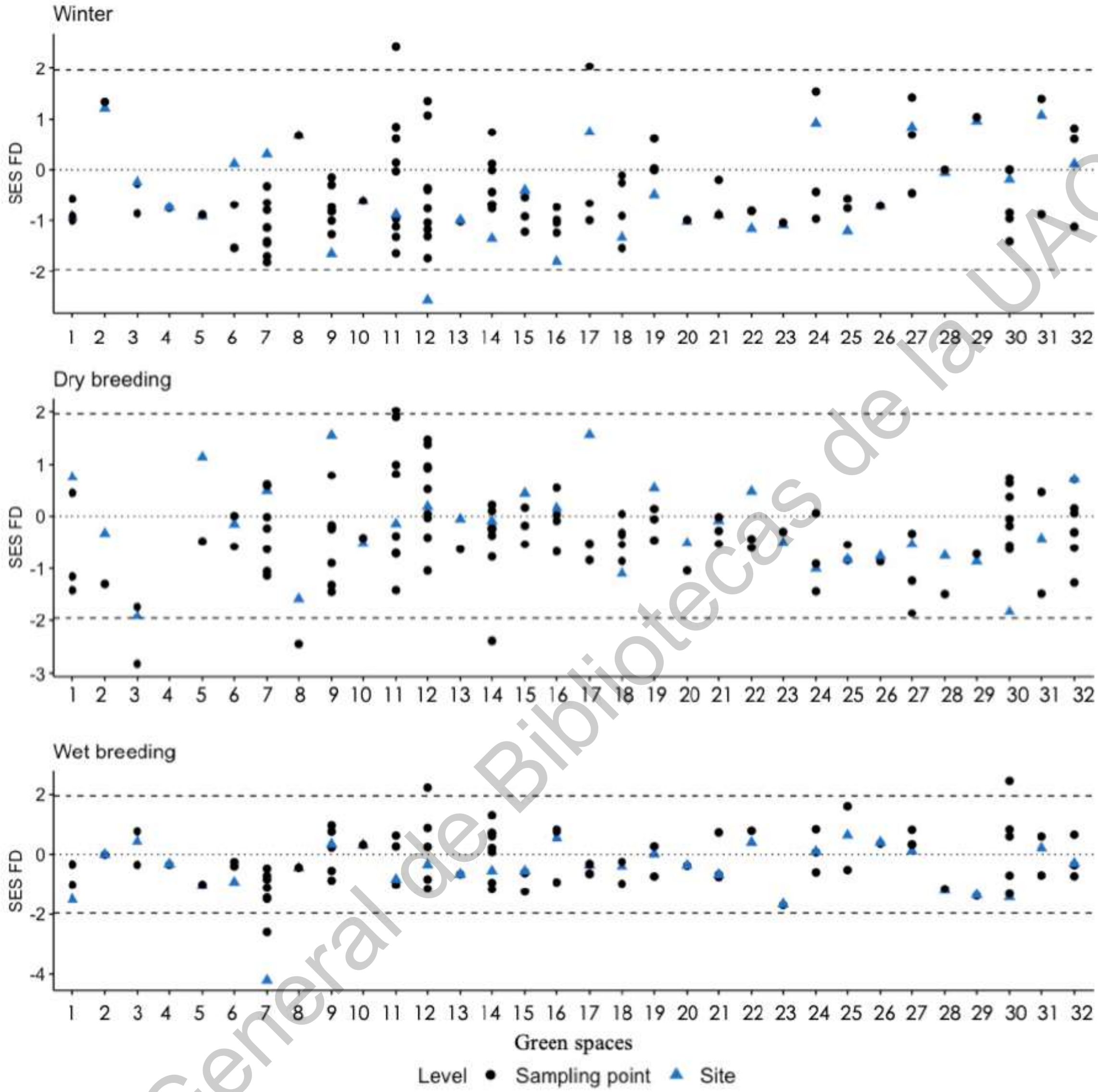

Fig. 6. Standardized effect sizes of Functional diversity during winter, dry breeding and wet breeding seasons at the site and sampling point levels. Values ≤ -1.96 (dashed line) indicate trait convergence while values ≥ 1.96 (dashed line) indicate trait divergence. Standardized effect sizes values = 0 are represented with a dotted line.

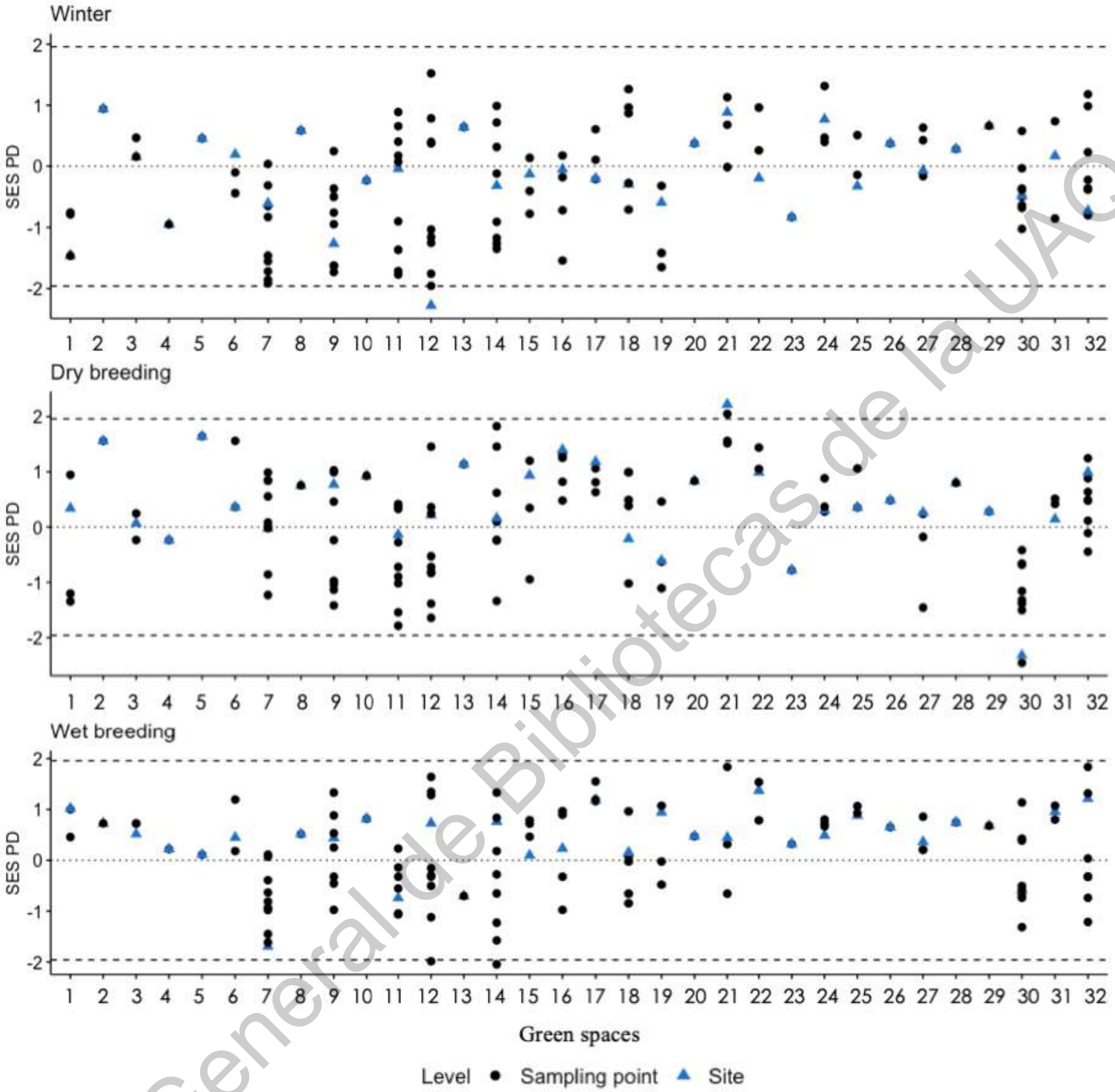

Fig. 7. Standardized effect sizes of Phylogenetic diversity during winter, dry breeding and wet breeding seasons at the site and sampling point levels. Values ≤ -1.96 (dashed line) indicate phylogenetic clustering while values ≥ 1.96 (dashed line) indicate phylogenetic over-dispersion. Standardized effect sizes values = 0 are represented with a dotted line.

Diversity predictors – site level

Surveyed green spaces showed considerable variation in their characteristics (Appendix 2). Green space area showed a strong significant correlation with tree richness ($R > 0.7$, $p < 0.05$) so we dropped the variable Area in the corresponding GLMs to account for collinearity among predictors. Satellite images presented cloud cover issues and hindered the DVF estimation for one green space during the dry and wet breeding seasons so this green space was discarded from analyses for the mentioned seasons. Overall, we found that the models including environmental information from just the green space were the most (n= 4) or the least (n= 9) supported models. Ranking and comparison of competing global models indicated that the addition of information about the densely vegetation fraction in the surroundings improved model fitting for some of the response variables (Table 3, Appendix 3). Models including information about the densely vegetation fraction in the surroundings explained up to 31% more variation than models not including this information. However, Akaike weights and evidence ratio indicated model uncertainty to some degree, with the weight of top-ranked models ranging from 0.32 to 0.76 with a mean = 0.50 (Appendix 3). Model uncertainty was less pronounced for models explaining taxonomic, phylogenetic, and functional diversity.

Results of model-averaging showed that the studied environmental variables significantly explained the variation in taxonomic, functional, and phylogenetic diversity, as well as the variation in SES of functional and phylogenetic diversity. Natural-averaged and zero-averaged models showed similar trends with the main difference being that zero-averaged models did not support significant predictors of SES phylogenetic diversity (Table 4, Appendix 4). Hereafter, we use natural-averaged models to describe the relationships between response variables and predictors.

A common feature of all the best-ranked models for species richness, phylogenetic diversity, and functional richness was that they all included tree species richness and densely-vegetated fraction (Table 4). Model-averaged coefficients indicated that the three bird diversity dimensions increased with tree richness in all seasons and that its effect was larger in summer (Fig. 8). Likewise, vegetation fraction affected positively taxonomic, phylogenetic, and functional diversity. On the contrary, during winter and the wet breeding season SES phylogenetic diversity increased as the densely-vegetated fraction decreased in the surroundings. This means that bird communities in green spaces tended to become phylogenetically clustered when the densely-vegetated fraction in the surroundings increased.

Table 3. Top-ranked global models from a set of four models proposed to explain diversity of birds in green spaces during winter (W), dry breeding (DB), and wet breeding (WB) seasons. Model were fitted for species richness, Faith's phylogenetic diversity, functional richness, and the standardized effect sizes of Faith's phylogenetic diversity and functional richness. Model weight (w_i) indicates the level of support of model *i*, considering the entire set of candidate models. DVF: Densely vegetated fraction. Densely-vegetated fraction. For more details about explanatory variables check Table 1.

	Candidate models	***k***	**w_i**	**R^2**
	Species richness			
W†	Habitat diversity + Isolation + Tree Richness + Shape + DVF$_{0}$ + DVF$_{50}$	7	0.74	67.7
DB	Habitat diversity + Isolation + Tree Richness + Shape + DVF$_{0}$ + DVF$_{100}$	7	0.43	60.0
W	Habitat diversity + Isolation + Tree Richness + Shape + DVF$_{0}$+ DVF$_{50}$	7	0.33	89.7
	Faith's phylogenetic diversity			
W	Habitat diversity + Isolation + Tree Richness + Shape + DVF$_{0}$ + DVF$_{50}$	8	0.48	56.3
DB	Habitat diversity + Isolation + Tree Richness + Shape + DVF$_{0}$ + DVF$_{100}$	8	0.32	51.3
WB	Habitat diversity + Isolation + Tree Richness + Shape + DVF$_{0}$ + DVF$_{50}$	8	0.61	82.1
	Standardized effect sizes of Faith's phylogenetic diversity			
W	Habitat diversity + Isolation + Tree Richness + Shape + DVF$_{0}$ + DVF$_{50}$	8	0.32	25.0
DB	Habitat diversity + Isolation + Tree Richness + Shape + DVF$_{0}$	7	0.56	15.0
WB	Habitat diversity + Isolation + Tree Richness + Shape + DVF$_{0}$ + DVF$_{200}$	8	0.61	49.8
	Functional richness			
W	Habitat diversity + Isolation + Tree Richness + Shape + DVF$_{0}$ + DVF$_{100}$	8	0.43	39.0
DB	Habitat diversity + Isolation + Tree Richness + Shape + DVF$_{0}$ + DVF$_{200}$	8	0.76	66.4
WB	Habitat diversity + Isolation + Tree Richness + Shape + DVF$_{0}$ + DVF$_{50}$	8	0.46	73.7
	Standardized effect sizes of functional richness			
W	Habitat diversity + Isolation + Tree Richness + Shape + DVF$_{0}$	7	0.64	22.1
DB	Habitat diversity + Isolation + Tree Richness + Shape + DVF$_{0}$	8	0.48	29.7
WB	Habitat diversity + Isolation + Tree Richness + Shape + DVF$_{0}$	7	0.42	14.3

† Over-dispersion was detected, so a quasi-GLM model was fitted.

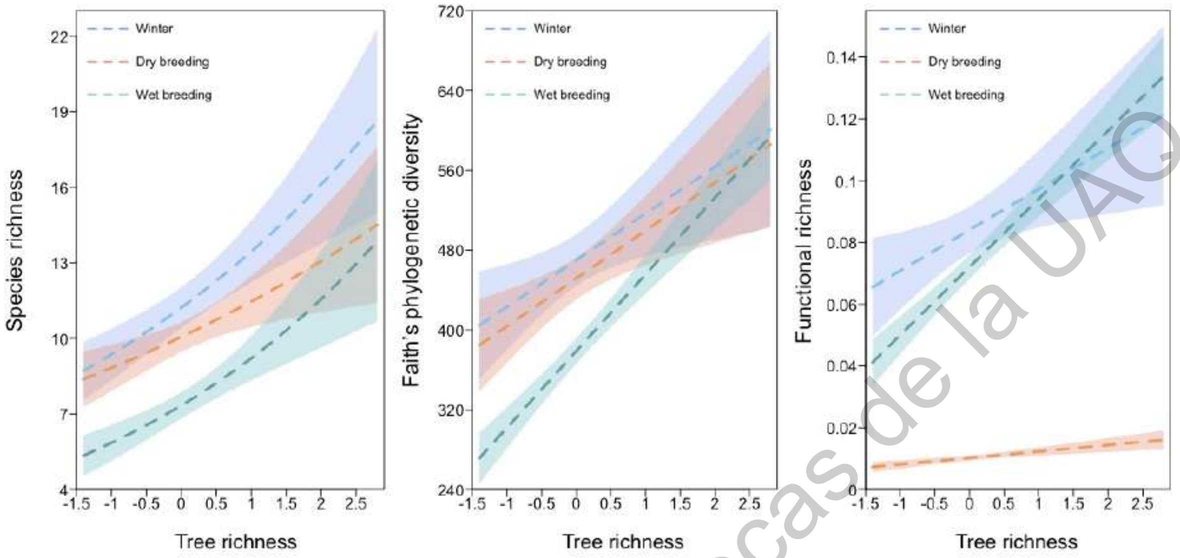

Fig. 8. Relationship between tree richness and different dimensions of bird diversity at the green space level. Taxonomic diversity was measured as species richness, phylogenetic diversity as Faith's phylogenetic diversity, and functional diversity as functional richness. Dashed lines represent predicted diversity values for each studied season based on generalized linear models. Shaded areas represent confidence intervals. Tree richness was scaled to mean = 0 and standard deviation = 1. Figure prepared with script found in Graham *et al*. (2019).

Model-averaging produced divergent results regarding habitat diversity effects on phylogenetic and functional diversity. Habitat diversity showed a positive relation with SES functional diversity while it showed a negative relation with SES phylogenetic diversity. This pattern was significant just for the wet breeding season. Furthermore, isolation showed a positive effect on SES functional diversity during winter. These results indicate that bird communities exhibit trait divergence under the following conditions: in isolated green spaces during winter or in green spaces with high habitat diversity during the wet breeding season. During the dry breeding season, SES functional diversity showed a positive relation with the densely-vegetated fraction measured within the limits of the green space (DVF_0).

Table 4. Natural-averaged estimates for predictors of taxonomic, phylogenetic, and functional diversity and the standardized effect sizes of phylogenetic diversity and functional richness at the site level. Models were fitted for winter, the dry breeding and the wet breeding seasons. Significant (bold) and marginally significant coefficients are shown. DVF: Densely vegetated fraction. Densely-vegetated fraction. For more details about explanatory variables check Table 1.

	DVF$_0$	Habitat Diversity	Isolation	Tree Richness	Shape	DVF$_{50}$	DVF$_{100}$	DVF$_{200}$
Season								
				Taxonomic diversity				
Winter				**0.183****		**0.364*****		
Dry breeding				**0.150***			**0.256****	
Wet breeding				**0.228****		**0.326****		
				Phylogenetic diversity				
Winter	-52.788			**59.913***		**129.25*****		
Dry breeding				**55.469***			**75.936***	
Wet breeding		**-42.898***		**76.884*****		**66.258****		
				Standardized effect size of Phylogenetic diversity				
Winter						**-0.322***		
Dry breeding					0.227			
Wet breeding								**-0.381****
				Functional diversity				
Winter				**0.017***			**0.020***	
Dry breeding				**0.002***				**0.004*****
Wet breeding				**0.021*****		**0.016****		
				Standardized effect size of Functional diversity				
Winter			**0.397***					
Dry breeding	**0.363***							
Wet breeding		**0.329**						

† Over-dispersion was detected, so a quasi-GLM model was fitted. *$P < 0.05$, **$P < 0.01$, ***$P < 0.001$

Predictor weights indicated that tree species richness was likely to be included in the best models of diversity measures across the studied seasons (Fig. 9). The rest of the predictors had a smaller likelihood to be included in the best models. In fact, predictors had relatively small weights in models fitted for SES values.

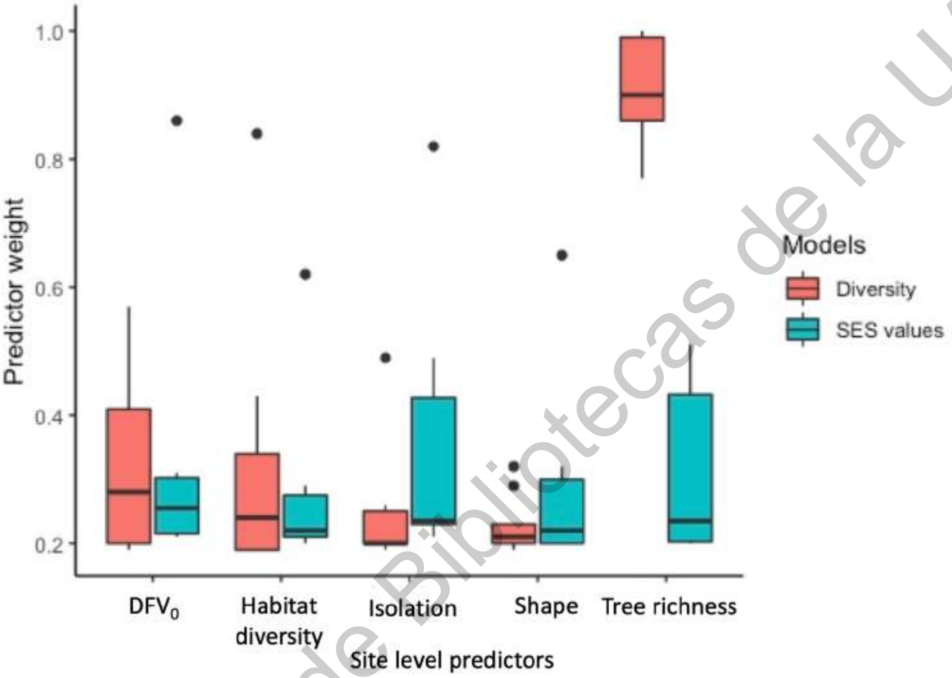

Fig. 9. Weights of predictors used in models to describe species richness, Faith's phylogenetic diversity, functional richness, and the standardized effect sizes of Faith's phylogenetic diversity and functional richness at the site level. Predictor weight (pw_i) indicates the relative importance of the predictor *i* and the likelihood that the predictor is included in the best model. DVF$_0$: Densely-vegetated fraction within the green space.

Diversity predictors – sampling point level

To account for collinearity among predictors, we eliminated from the analyses the following variables due to a strong correlation with other variables: bare ground and noise from the winter data subset, and bare ground from the dry breeding data subset. Hence, the number of predictors retained in the models differed among seasons. Natural-averaged and zero-averaged models showed equivalent trends but few relationships remained significant (Table 5, Appendix 5). Model-averaging showed that response variables were explained by different predictors during winter, the dry breeding and the wet breeding seasons (Table 5). Among the vegetation descriptors, tree species richness and tree height showed a positive relationship with taxonomic and phylogenetic diversity.

Ground cover types showed different effects on the response variables. While shrub cover had a positive effect on taxonomic, phylogenetic, and functional diversity during winter, tree cover had a negative effect on phylogenetic and functional diversity and their SES values. Results revealed that the variables describing the disturbance regime had a significant effect on bird diversity dimensions. For example, model averaging identified a significant positive effect of noise on taxonomic diversity, phylogenetic diversity, and phylogenetic structure. Additionally, the equipment in sampling points had a positive effect on the phylogenetic structure of bird communities, while the same variable had a negative effect on their functional structure.

The relative importance of predictors differed between models fitted for diversity measures and models fitted for SES values (Fig. 10). Akaike weights indicate that noise and tree richness were the most likely predictors to be included in models explaining variation in diversity. The equipment of sampling points was the most supported variable in models fitted for SES values.

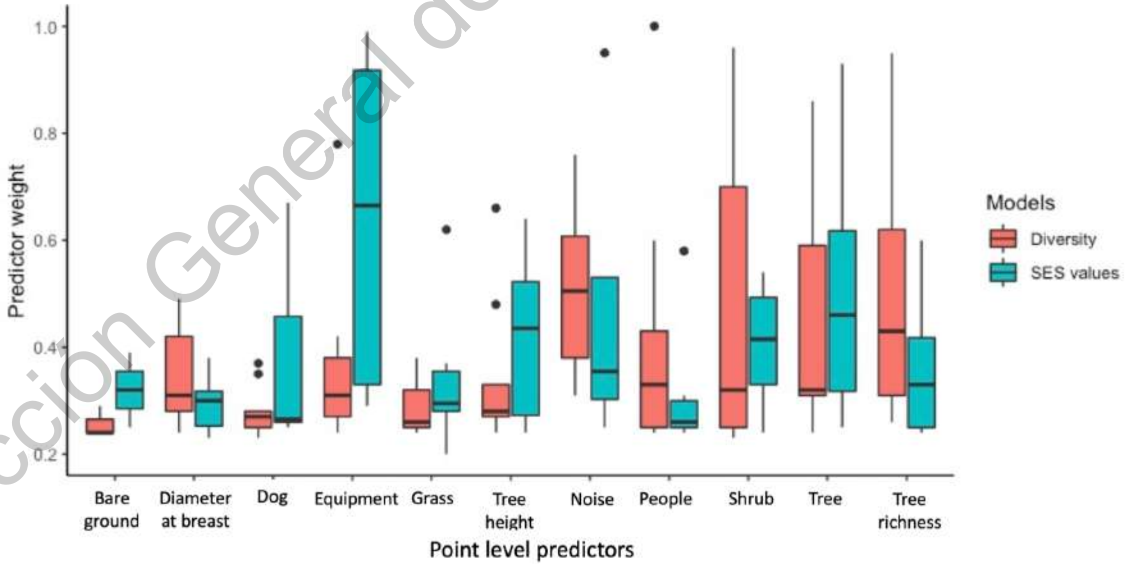

Fig. 10. Weights of predictors used in models to describe species richness, Faith's phylogenetic diversity, functional richness, and the standardized effect sizes of Faith's phylogenetic diversity and functional richness at the sampling point level. Predictor weight (pw$_i$) indicates the relative importance of the predictor *i* and the likelihood that the predictor is included in the best model. Variables are described in Table 1.

Table 5. Natural-averaged estimates for predictors of taxonomic, phylogenetic, and functional diversity and the standardized effect sizes of phylogenetic diversity and functional richness at the sampling-point level. Models were fitted for winter, the dry breeding and the wet breeding seasons. Significant (bold) and marginally significant coefficients are shown.

	Diameter at breast	Tree height	Tree richness	Bare ground	Grass	Shrub	Tree	Equipment	Dog	People	Noise
Season											
Taxonomic diversity											
Winter		0.097		NA		**0.100***					NA
Dry breeding				NA						-0.093	**0.110***
Wet breeding[†]			0.115								
Phylogenetic diversity											
Winter			22.527	NA		**33.811****					NA
Dry breeding			20.251	NA			-19.794	**34.302***			25.299
Wet breeding			**37.537****								23.007
Standardized effect size of Phylogenetic diversity											
Winter				NA				**0.325****			NA
Dry breeding		-0.187	0.174	NA			-0.178	**0.338****			
Wet breeding									0.189		**0.301****
Functional diversity											
Winter				NA		**0.010****					NA
Dry breeding				NA			-0.005			**-0.012*****	
Wet breeding							**-0.010***				
Standardized effect size of Functional diversity											
Winter				NA	-0.181						NA
Dry breeding				NA				**-0.383*****		-0.155	
Wet breeding							**-0.306****				

[†] Over-dispersion was detected, so a negative binomial GLMM model was fitted. *$P < 0.05$, **$P < 0.01$, ***$P < 0.001$

Discussion

Diversity and structure of bird communities in green spaces of Mexico City

Green spaces of Mexico City surveyed in this study harbored 34% of the land bird species reported in the state (Meléndez-Herrada *et al*., 2016). Taxonomic, phylogenetic and functional diversity differed among green spaces indicating that these sites possess different value for urban bird conservation. As expected, bird diversity varied seasonally with the highest taxonomic, phylogenetic and functional diversity observed during winter. It has been suggested that seasonal differences in urban bird assemblages can be explained by imperfect detection and seasonal use of alternative habitats (Murgui, 2010). In fact, detectability could have varied among seasons and habitats (Iknayan *et al*., 2014), but we used a 25-m point count to reduces detectability differences (Hutto *et al*., 1986). Therefore, we assume these differences were not main drivers of the detected seasonal variation.

We suggest that the presence of wintering species in urban green spaces during winter is the main driver of seasonal differences in bird diversity dimensions. A close inspection of species incidence across the seasons (Appendix 1) reveals that during winter, functionally and phylogenetically distinctive species were present in surveyed green spaces. For example, American Krestel and Plumbeous Vireo are respectively the only members belonging to order Falconiformes and family Vireonidae that were detected in this study and were present only during winter. Hence, our study supports the importance of urban green spaces for migratory bird conservation (Dale, 2018) and highlights the contribution of migratory species to bird diversity in urban settings.

Our results showed that reduced species richness translated into reduced phylogenetic and functional diversity. However, coupling between taxonomic and functional diversity was not as strong as coupling between taxonomic and phylogenetic diversity. In the case of functional diversity, a correlation value ≈1 has been interpreted as an indicator of functional redundancy of assemblages (Petchey *et al.*, 2007). Following this criterion, our findings suggest that the absence of a species from a green space does not necessarily imply the loss of ecological functions since functionally similar species may remain in the green space.

The relationships among taxonomic, phylogenetic and functional diversity varied across the studied seasons and all were stronger during the wet breeding season, except for the relationship between phylogenetic and functional diversity at the sampling-point level. This could indicate that the city pool of species (*sensu* Aronson *et al.*, 2016) from which species colonized green spaces were less functionally and phylogenetically redundant during the wet breeding season. In the study area, woodpeckers (Picidae), flycatchers (Tyrannidae), kinglets (Regulidae), thrushes (Turdidae), new world sparrows (Passeellidae), new world warblers (Parulidae), and tanagers (Cardinalidae) are part of the resident breeding assemblages which during winter, contain additional either migratory or transient species of these families.

An alternative and complementary explanation would be that for relatively small numbers of species, the addition of "unique" clades or traits is more likely. A saturation effect has been reported for phylogenetic and functional diversity of bird assemblages in farmlands (García-Navas and Thuiller, 2020). These authors found that the standardized effect size of both phylogenetic and functional tended to decrease when species richness reached a threshold (near ≈35 species). In this study, at the site level, functional richness seemed to reach and asymptote above species richness ≈20. Regarding phylogenetic diversity, in this study, our results did not suggest an asymptotic relationship with species richness.

The relationship among response variables was equivalent when comparing site-level correlations with sampling-point level correlations. Overall, relations had the same direction but they were weaker at sampling points. For example, the relationship between species richness and functional diversity was always positive, but it was weaker at the sampling-point level. These differences may be the result of a larger variation in the functional diversity of local species pools (*sensu* Aronson *et al.*, 2016). Once species have been sorted regionally and by the city characteristics, at smaller scales, for example districts, neighborhoods or green spaces, environmental heterogeneous habitats could lead to the functional differentiation of local avifaunas, and this may translate into different species richness-functional diversity relationships. These findings are important for future studies: although taxonomic diversity cannot capture all the evolutionary history variation or phenotypic variation (Ibáñez-Álamo *et al.*, 2020), it can be used in specific seasons of the year as a surrogate for phylogenetic diversity, and to a lesser extent, as a surrogate for functional diversity.

Our results did not support our second hypothesis: the urban environment would filter out some species and consequently, bird assemblages of the studied green spaces would show a clustered phylogenetic and functional structure. We found a predominantly random functional and phylogenetic structure of bird communities in green spaces of Mexico City. That is, there was no evidence to conclude that the environmental conditions of the most urbanized green spaces posed a filter to traits or clades. However, values lower than the expected ones were observed in two green spaces. These sites resembled natural habitats and were located far from downtown and close to the natural forested areas of the city. Based on this, we propose that species that prefer open habitats within cities may react to forests as local environmental filters, as has been suggested for forest-farmland and grassland-forest plantation systems (Jacoboski *et al*., 2019; García-Navas and Thuiller, 2020). Likewise, species that need large forest patches may be filtered out from small isolated green spaces. However, compensatory dynamics may be occurring (Supp and Ernst, 2014), by which losses of traits and clades in small isolated green spaces are probably compensated by the introduction of exotic species, a mechanism not occurring in large and less landscaped green spaces.

Predictors: site level

Densely-vegetated fraction and tree species richness were important habitat factors for taxonomic, phylogenetic, and functional diversity of green spaces. Findings are consistent with previous reports about the positive effect of these factors on different dimensions of bird diversity (Leveau *et al*., 2020). We recall that tree species richness functioned as a surrogate of green space area, which is one of the most influential factors of biodiversity in urban habitat patches (Beninde *et al*., 2015; La Sorte *et al*., 2020). Therefore, species-area relationships may underlie the observed tree richness effect, so we recommend a cautious interpretation of the relative importance of tree richness.

The use of an information theoretic approach allowed us to evaluate alternative biological hypothesis about the role of dense vegetation within green spaces and in their close surroundings. Overall, models including information about dense vegetation in the examined buffers were better ranked. We want to highlight that the three dimensions of bird diversity were better explained when estimations of the densely-vegetated fraction in the surroundings were considered. This improvement in model fit was observed across the seasons. These results support previous evidence on the influence of landscape urbanization on bird diversity of green spaces (Carbó-Ramírez and Zuria, 2011) and confirm that bird diversity of green spaces is favored by less paved surroundings (Schütz and Schulze, 2015).

Our results showed that phylogenetic diversity decreased as habitat diversity increased during the wet breeding season. This was an unexpected result based on the fact that habitat heterogeneity can positively affect species richness (Loss *et al.*, 2009) and that we detected a positive relationship between species richness and phylogenetic diversity. We also want to highlight that models fitted for SES values explained a relatively small fraction of variation. We suggest that to understand the magnitude of the departure from randomness in phylogenetic and functional diversity, different predictors need to be considered.

In accordance with existing evidence (La Sorte *et al*., 2020), isolation and green space shape did not have a significant effect on taxonomic, functional, nor phylogenetic diversity. However, the standardized effect size of functional diversity was significantly affected by isolation during winter. In other words, during winter, greater green space isolation enriches the functional variability of bird assemblages.

Predictors: sampling point level

Several habitat variables at the sampling point turned out to be relevant factors for taxonomic, phylogenetic, and functional diversity. Habitat features associated with vegetation have been identified as relevant environmental drivers of bird diversity in urban green spaces (Beninde *et al.*, 2015; Morelli *et al.*, 2017) and our results confirm this. Similarly, human-related factors such as noise levels and equipment revealed as significant factors driving the variation of different dimensions of bird diversity. In the case of the variable equipment, such artificial structures may offset the disappearance of natural structural elements in the more landscaped places. In this way, species that benefit from these structures, such as the Inca Dove (Ortega-Álvarez and MacGregor-Fors, 2010), can persist and thrive in transformed sites. Our results suggest that this possible mechanism may have contrasting effects on phylogenetic and functional diversity. While the evolutionary history of the assemblage can be enriched by species that exploit artificial structures such as House Sparrow, Rock Pigeon, and Inca Dove, the functional diversity may not, due to trait convergence since these species share common traits such as ground-foraging and seed-based diet.

We found that noise levels correlated positively with taxonomic diversity and phylogenetic SES values. This is an unexpected result considering that previous works have reported deleterious effects of noise on bird richness and abundance in urban habitats (Carbó-Ramírez and Zuria, 2011; Barbosa *et al*., 2020; Planillo *et al*., 2021). However, there is also evidence for non-significant effects of noise on species richness in urban parks (Yang *et al*., 2020) and for differences in species sensitivity to noise (Kunc and Schmidt, 2021). To unravel the effects of noise on the different dimensions of bird diversity in urban parks, further studies could benefit from controlling variation in other relevant factors.

Finally, and in agreement with previous works (Levau and Leveau, 2016; La Sorte *et al*., 2020), our results showed that environment-diversity relationships varied across seasons. We propose that temporal changes in species urban tolerance (Callaghan *et al*., 2021), anthropic related mortality (Bracey *et al*., 2016), anthropogenic food availability (Stofberg *et al*., 2019) could be mechanisms simultaneously operating and could underlie the temporal variation. A closer examination of the occurrence of specific clades, traits, and species under specific environmental conditions is needed. So, future studies should consider both spatial and temporal variation of environmental conditions together with temporal differences in habitat requirements of species.

CONCLUSIONES GENERALES

El presente trabajo contribuye al conocimiento sobre la diversidad de aves en áreas verdes urbanas, tanto en el contexto mexicano como en el contexto internacional. A nivel nacional, se logró compendiar la información existente al inicio de esta investigación y se identificaron líneas de investigación que no habían sido exploradas aún. Los hallazgos de esta revisión denotan que, si bien el estudio de las aves urbanas y su ecología en México ha recibido la atención de la comunidad académica, las preguntas planteadas hasta esa fecha, se habían limitado a un número reducido de temas. De igual manera, los resultados de la revisión permitieron identificar que gran parte de los estudios se han realizado en unas cuantas regiones del país lo que implica que aún desconocemos las respuestas de la gran diversidad de avifauna nacional a la urbanización.

A partir de la revisión, se condujo un estudio en la ciudad de Mérida, para cubrir un vacío de información detectado. Los resultados señalaron que el efecto de las características de las áreas verdes es dependiente de las especies y que esto tiene implicaciones para la distribución y abundancia de las especies en las áreas verdes. Resalta que este trabajo mostró que las especies vegetales nativas y exóticas influyen de manera diferente a las especies.

Este trabajo demostró que la matriz que rodea a las áreas verdes juega un papel ecológico relevante pues influye de manera significativa en la diversidad de aves. Este puede ser un hecho del cual se pueda tomar ventaja para mantener o incrementar la diversidad de aves en las áreas verdes. Al igual que la matriz, en su interior son múltiples los factores de las áreas verdes con un efecto significativo. Otro aporte valioso del trabajo fue utilizar medidas de diversidad complementarias a la diversidad taxonómica, una contribución valiosa para una zona urbana tan estudiada como la Ciudad de México. Al proceder así, encontramos que las diferentes dimensiones de la diversidad son afectadas por diferentes factores ambientales. De igual manera, mostramos que las relaciones diversidad-ambiente no son consistentes entre las temporadas. Ambos hallazgos tienen implicaciones para la conservación de aves en las áreas verdes urbanas.

En conclusión, los estudios sobre la ecología de aves en áreas verdes urbanas deberían incorporar diferentes medidas de diversidad, tener una perspectiva temporal y considerar factores de las áreas verdes a más de una escala espacial para brindar un panorama más completo del fenómeno ecológico en cuestión: la diversidad de aves en áreas verdes urbanas y sus determinantes.

REFERENCIAS

Acuña-Muñoz, J. D. (2014). *Avifauna de la FES Iztacala, UNAM. Estudio comparativo.* (Tesis de licenciatura), Universidad Nacional Autónoma de México, Tlalnepantla, México.

Alberti, M. (2008). *Advances in Urban Ecology: Integrating Humans and Ecological Processes in Urban Ecosystems*. New York: Springer.

Alexander, J., Smith, D. A. E., Smith, Y. C. E., & Downs, C. T. (2019). Drivers of fine-scale avian functional diversity with changing land use: an assessment of the effects of eco-estate housing development and management. *Landscape Ecology, 34*(3), 537-549. doi:10.1007/s10980-019-00786-y

Almazán-Núñez, C., & Hinterholzer-Rodríguez, A. (2010). Dinámica temporal de la avifauna en un parque urbano de la ciudad de Puebla, México. *Huitzil, 11*(1), 26-34.

Almeida, B. D., Green, A. J., Sebastian-Gonzalez, E., & dos Anjos, L. (2018). Comparing species richness, functional diversity and functional composition of waterbird communities along environmental gradients in the neotropics. *Plos One, 13*(7). doi:10.1371/journal.pone.0200959

Almejo, R., García, J., & Benítez, I. (2014). La urbanización en México 2010-2030: un esbozo de los retos y oportunidades asociados al crecimiento urbano y regional. En CONAPO (Ed.), *La situación demográfica de México 2014* (pp. 139-164). México: CONAPO.

Arenas, S. (2004). *Distribucion y fenologia de la avifauna del Ajusco medio y del Pedregal de San Ángel, Distrito Federal México.* (Tesis de licenciatura), Universidad Nacional Autónoma de México, Distrito Federal, México.

Aronson, M. F. J., La Sorte, F. A., Nilon, C. H., Katti, M., Goddard, M. A., Lepczyk, C. A., . . . Winter, M. (2014). A global analysis of the impacts of urbanization on bird and plant diversity reveals key anthropogenic drivers. *Proceedings of the Royal Society B-Biological Sciences, 281*(1780). doi:10.1098/rspb.2013.3330

Aronson, M. F. J., Nilon, C. H., Lepczyk, C. A., Parker, T. S., Warren, P. S., Cilliers, S. S., . . . Zipperer, W. (2016). Hierarchical filters determine community assembly of urban species pools. *Ecology, 97*(11), 2952-2963. doi:10.1002/ecy.1535

Barbosa, K. V. D. C., Rodewald, A. D., Ribeiro, M. C., & Jahn, A. E. (2020). Noise level and water distance drive resident and migratory bird species richness within a Neotropical megacity. *Landscape and Urban Planning, 197*. doi:10.1016/j.landurbplan.2020.103769

Barth, J. B., FitzGibbon, I. S., & Wilson, S. R. (2015). New urban developments that retain more remnant trees have greater bird diversity. *Landscape and Urban Planning, 136*, 122-129. doi: 10.1016/j.landurbplan.2014.11.003

Bartón, K., (2020). MuMIn: Multi-Model Inference. R package version 1.43.17. https://CRAN.R-project.org/package=MuMIn

Bates, D., Mächler, M., Bolker, B., & Walker, S. (2015). Fitting Linear Mixed-Effects Models Using lme4. *Journal of Statistical Software, 67*(1), 1-48. doi:10.18637/jss.v067.i01

Beninde, J., Veith, M., & Hochkirch, A. (2015). Biodiversity in cities needs space: a meta-analysis of factors determining intra-urban biodiversity variation. *Ecology Letters, 18*(6), 581-592. doi:10.1111/ele.12427

Benitez, J., Pizarro, J. C., Blazina, A. P., & Lencinas, M. V. (2021). Response of bird communities to native forest urbanization in one of the southernmost city of the world. *Urban Forestry & Urban Greening, 58*, 126887. doi:10.1016/j.ufug.2020.126887

Bernhardt-Romermann, M., Romermann, C., Nuske, R., Parth, A., Klotz, S., Schmidt, W., & Stadler, J. (2008). On the identification of the most suitable traits for plant functional trait analyses. *Oikos, 117*(10), 1533-1541. doi:10.1111/j.2008.0030-1299.16776.x

Bhakti, T., Goulart, F., de Azevedo, C. S., & Antonini, Y. (2018). Does scale matter? The influence of three- level spatial scales on forest bird occurrence in a tropical landscape. *Plos One, 13*(6). doi:10.1371/journal.pone.0198732

Bibby, C. J., Burgess, N. D., & Hill, D. A. (1993). *Bird Census Techniques*. San Diego: Academic Press.

BirdLife International (2018). Bird species distribution maps of the world. Version 6.0. Recuperado de http://datazone.birdlife.org/species/requestdis

Blair, R. B. (1996). Land Use and Avian Species Diversity Along an Urban Gradient. *Ecological Applications, 6*(2), 506-519. doi:10.2307/2269387

Blair, R. B. (1999). Birds and butterflies along an urban gradient: surrogate taxa for assessing biodiversity? *Ecological Applications, 9*(1), 164-170. doi:10.1890/1051-0761(1999)009[0164:BABAAU]2.0.CO;2

Borcard, D., Gillet, F., & Legendre, P. (2018). *Numerical Ecology with R*. New York: Springer.

Bovo, A. A. A., Ferraz, K., Magioli, M., Alexandrino, E. R., Hasui, E., Ribeiro, M. C., & Tobias, J. A. (2018). Habitat fragmentation narrows the distribution of avian functional traits associated with seed dispersal in tropical forest. *Perspectives in Ecology and Conservation, 16*(2), 90-96. doi:10.1016/j.pecon.2018.03.004

Bracey, A. M., Etterson, M. A., Niemi, G. J., & Green, R. F. (2016). Variation in bird-window colission mortality and scavenging rates within an urban landscape. *Wilson Journal of Ornithology, 128*(2), 355-367. doi:10.1676/wils-128-02-355-367.1

Burnham, K., & Anderson, D. R. (2002). *Model Selection and Multimodel Inference: A Practical Information-Theoretic Approach, 2nd edn.* New York: Springer.

Buzo-Franco, D., & Hernández-Satín, L. (2004). *Dinámica espacial y temporal de la comunidad de aves en los parques urbanos de Puebla y su entorno.* (Tesis de licenciatura), Universidad de las Américas Puebla, Puebla, México.

Buzo-Franco, D., L. Hernández-Santín, C. Bonache-Regidor y J. A. González-Oreja, (2005). Dinámica espacial de la comunidad de aves en los parques urbanos de Puebla y su entorno. Pp. 95. En XVIII Congreso Nacional de Zoología. Sociedad Mexicana de Zoología, A. C. Monterrey.

Callaghan, C. T., Cornwell, W. K., Poore, A. G. B., Benedetti, Y., & Morelli, F. (2021). Urban tolerance of birds changes throughout the full annual cycle. *Journal of Biogeography*. doi:10.1111/jbi.14093

Calmé, S., MacKinnon-H, B., Leyequién, E., & Escalona-Segura, G. (2015). Birds. In G. A. Islebe, S. Calmé, J. L. León-Cortés, & B. Schmook (Eds.), *Biodiversity and Conservation of the Yucatán Peninsula* (pp. 295-332): Springer.

Canedoli, C., Orioli, V., Padoa-Schioppa, E., Bani, L., & Dondina, O. (2017). Temporal Variation of Ecological Factors Affecting Bird Species Richness in Urban and Peri-Urban Forests in a Changing Environment: A Case Study from Milan (Northern Italy). *Forests, 8*(12). doi:10.3390/f8120507

Carbó-Ramírez, P. y G. López-Ortega, (2005). Fluctuaciones estacionales en la diversidad de aves en el campus de la Universidad Autónoma Metropolitana Iztapalapa. Pp. 143. En XVIII Congreso Nacional de Zoología. Sociedad Mexicana de Zoología, A. C. Monterrey.

Carbó-Ramírez, P., & Zuria, I. (2011). The value of small urban greenspaces for birds in a Mexican city. *Landscape and Urban Planning, 100*(3), 213-222. doi:10.1016/j.landurbplan.2010.12.008

Carbó-Ramírez, P., Zuria, I., & Romero-González, M. P. (2011). Riqueza, abundancia y dinámica espacio-temporal de la comunidad de aves de Ciudad Universitaria, Universidad Autónoma del Estado de Hidalgo, Pachuca, México *El canto del Centzontle, 2*(1), 29-47.

Cárdenas-Carmona, D. U. (2014). *Los Parques Funerarios Un Refugio Importante Para Las Aves en la Ciudad de Guadalajara y su Zona Conurbada.* (Tesis de licenciatura), Universidad de Guadalajara, Guadalajara, México.

Carvajal-Castro, J. D., Ospina-L, A. M., Toro-López, Y., Pulido-G, A., Cabrera-Casas, L. X., Guerrero-Peláez, S., . . . Vargas-Salinas, F. (2019). Birds vs bricks: Patterns of species diversity in response to urbanization in a Neotropical Andean city. *Plos One, 14*(6), e0218775. doi:10.1371/journal.pone.0218775

Castaño-Villa, G. J., Santisteban-Arenas, R., Hoyos-Jaramillo, A., Estévez-Varón, J. V., & Fontúrbel, F. E. (2019). Foraging behavioural traits of tropical insectivorous birds lead to dissimilar communities in contrasting forest habitats. *Wildlife Biology, 2019*(1), 1-6, 6.

Castro-Torreblanca, M., & Blancas Calva, E. (2014). Aves de Ciudad Universitaria campus Sur de la Universidad Autónoma de Guerrero, Chilpancingo, Guerrero, México. *Huitzil, 15*(2), 82-92.

Charre, G. M. (2013). *Distribución, cambios en algunas pautas conductuales y preferencias de forrajeo de aves migratorias insectívoras en parques de la Ciudad de México.* (Tesis de doctorado), Universidad Autónoma Metropolitana, Distrito Federal, México.

Charre, G. M., Hurtado, J. A. Z., Neve, G., Ponce-Mendoza, A., & Corcuera, P. (2013). Relationship Between Habitat Traits And Bird Diversity And Composition In Selected Urban Green Areas Of Mexico City. *Ornitologia Neotropical, 24*(3), 279-297.

Chave, J. (2004). Neutral theory and community ecology. *Ecology Letters, 7*(3), 241-253. doi:10.1111/j.1461-0248.2003.00566.x

Cisneros, L. M., Fagan, M. E., & Willig, M. R. (2015). Effects of human-modified landscapes on taxonomic, functional and phylogenetic dimensions of bat biodiversity. *Diversity and Distributions, 21*(5), 523-533. doi:10.1111/ddi.12277

Colwell, R. K., (2013). EstimateS: Statistical estimation of species richness and shared species from samples. Version 9. User's Guide and application published at: http://purl.oclc.org/estimates.

Comisión Nacional para el Conocimiento y Uso de la Biodiversidad (CONABIO), (1997). Provincias biogeográficas de México. Escala 1:4000000. Comisión Nacional para el Conocimiento y Uso de la Biodiversidad, México, D. F.

Comisión Nacional para el Conocimiento y Uso de la Biodiversidad (CONABIO), (2015). Avesmx.net. Última consulta 15 de octubre de 2015. Recuperado de: http://avesmx.conabio.gob.mx/busca_region.html.

Consejo Nacional de Población (CONAPO), (2015). Delimitación de las zonas metropolitanas de México. Recuperado de: https://www.gob.mx/conapo/documentos/delimitacion-de-las-zonas-metropolitanas-de-mexico-2015

Corey, S. S., Christopher, E. M., Jason, D. R., & Burchell, M. R. (2012). Influence of Patch Size and Shape on Occupancy by Shrubland Birds. *The Condor, 114*(2), 268-278. doi:10.1525/cond.2012.110107

Cornelis, J., & Hermy, M. (2004). Biodiversity relationships in urban and suburban parks in Flanders. *Landscape and Urban Planning, 69*, 385-401. doi:10.1016/j.landurbplan.2003.10.038

Cristaldi, M. A., Giraudo, A. R., Arzamendia, V., Bellini, G. P., & Claus, J. (2017). Urbanization impacts on the trophic guild composition of bird communities. *Journal of Natural History, 51*(39-40), 2385-2404. doi:10.1080/00222933.2017.1371803

Dale, S. (2018). Urban bird community composition influenced by size of urban green spaces, presence of native forest, and urbanization. *Urban Ecosystems, 21*(1), 1-14. doi:10.1007/s11252-017-0706-x

Davis, A., Major, R. E., & Taylor, C. E. (2015). The association between nectar availability and nectarivore density in urban and natural environments. *Urban Ecosystems, 18*(2), 503-515. doi:10.1007/s11252-014-0417-5

del Hoyo, J., Elliot, A., Sargatal, J., Christie, D. A., & de Juana, E. (2018). Handbook of the Birds of the World Alive. Recuperado de http://www.hbw.com/

Delaney, K. S., Riley, S. P. D., & Fisher, R. N. (2010). A Rapid, Strong, and Convergent Genetic Response to Urban Habitat Fragmentation in Four Divergent and Widespread Vertebrates. *Plos One, 5*(9). doi:10.1371/journal.pone.0012767

Deppe, J. L., & Rotenberry, J. T. (2005). Temporal patterns in fall migrant communities in Yucatan, Mexico. *Condor, 107*(2), 228-243. doi:10.1650/7804

Devictor, V., Mouillot, D., Meynard, C., Jiguet, F., Thuiller, W., & Mouquet, N. (2010). Spatial mismatch and congruence between taxonomic, phylogenetic and functional diversity: the need for integrative conservation strategies in a changing world. *Ecology Letters, 13*(8), 1030-1040. doi:10.1111/j.1461-0248.2010.01493.x

Di Pietro, S., Mantoni, C., & Fattorini, S. (2020). Influence of urbanization on the avian species-area relationship: insights from the breeding birds of Rome. *Urban Ecosystems*. doi:10.1007/s11252-020-01081-4

Díaz, T. (2008). *Distribución espacio-temporal de la avifauna en la zona árida y zona templada del Jardín Botánico del Instituto de Biología y los viveros de la Unidad de Seminarios Ignacio Chávez.* (Tesis de licenciatura), Universidad Nacional Autónoma de México, México.

Dirzo, R., Young, H. S., Galetti, M., Ceballos, G., Isaac, N. J. B., & Collen, B. (2014). Defaunation in the Anthropocene. *Science, 345*(6195), 401-406. doi:10.1126/science.1251817

Domínguez, R., J. Chablé y J. S. Flores. 2011. Aves y huertos familiares en la zona periurbana de la ciudad de Mérida, Yucatán. Pp. 18. En XI Congreso para el estudio y conservación de las aves en México (CECAM). Sociedad para el Estudio y Conservación de las Aves en México A. C. Mazatlán.

Dray, S., & Dufour, A. B. (2007). The ade4 package: Implementing the duality diagram for ecologists. *Journal of Statistical Software, 22*(4), 1-20.

Dray, S., & Legendre, P. (2008). Testing the Species Traits-Environment Relationships: the Fourth-Corner Problem Revisited. *Ecology, 89*(12), 3400-3412. doi:10.1890/08-0349.1

Dray, S., Choler, P., Doledec, S., Peres-Neto, P. R., Thuiller, W., Pavoine, S., & ter Braak, C. J. F. (2014). Combining the fourth-corner and the RLQ methods for assessing trait responses to environmental variation. *Ecology, 95*(1), 14-21.

Duarte, M. T. (2001). *Caracterización de la comunidad de aves de la UNAM Campus Iztacala.* (Tesis de licenciatura). Universidad Nacional Autónoma de México, Tlalnepantla de Baz, México.

Dunford, W., & Freemark, K. (2005). Matrix matters: Effects of surrounding land uses on forest birds near Ottawa, Canada. *Landscape Ecology, 20*(5), 497-511. doi:10.1007/s10980-004-5650-5

Evans, K. L., Newson, S. E., & Gaston, K. J. (2009). Habitat influences on urban avian assemblages. *Ibis, 151*(1), 19-39. doi:10.1111/j.1474-919X.2008.00898.x

Faith, D. P. (1992). Conservation evaluation and phylogenetic diversity. *Biological Conservation, 61*(1), 1-10. doi:10.1016/0006-3207(92)91201-3

Faith, D. P. (1994). Phylogenetic Pattern and the Quantification of Organismal Biodiversity. *Philosophical Transactions: Biological Sciences, 345*(1311), 45-58.

Faith, D. P. (2002). Quantifying Biodiversity: a Phylogenetic Perspective. *Conservation Biology, 16*(1), 248-252. doi:10.1046/j.1523-1739.2002.00503.x

Fernandez-Juricic, E. (2000). Bird community composition patterns in urban parks of Madrid: The role of age, size and isolation. *Ecological Research, 15*(4), 373-383. doi:10.1046/j.1440-1703.2000.00358.x

Fernandez-Juricic, E., & Jokimaki, J. (2001). A habitat island approach to conserving birds in urban landscapes: case studies from southern and northern Europe. *Biodiversity and Conservation, 10*(12), 2023-2043.

Fernandez-Juricic, E. (2002). Can human disturbance promote nestedness? A case study with breeding birds in urban habitat fragments. *Oecologia, 131*(2), 269-278. doi:10.1007/s00442-002-0883-y

Filippi-Codaccioni, O., Clobert, J., & Julliard, R. (2009). Urbanisation effects on the functional diversity of avian agricultural communities. *Acta Oecologica-International Journal of Ecology, 35*(5), 705-710. doi:10.1016/j.actao.2009.07.003

Fragkias, M., Güneralp, B., Seto, K. C., & Goodness, J. (2013). A Synthesis of Global Urbanization Projections. In T. Elmqvist, M. Fragkias, J. Goodness, B. Güneralp, P. J. Marcotullio, R. I. McDonald, S. Parnell, M. Schewenius, M. Sendstad, K. C. Seto, & C. Wilkinson (Eds.), *Urbanization, Biodiversity and Ecosystem Services: Challenges* *and Opportunities: A Global Assessment* (pp. 409-435). Dordrecht: Springer Netherlands.

Frishkoff, L. O., Karp, D. S., M'Gonigle, L. K., Mendenhall, C. D., Zook, J., Kremen, C., . . . Daily, G. C. (2014). Loss of avian phylogenetic diversity in neotropical agricultural systems. *Science, 345*(6202), 1343-1346. doi:10.1126/science.1254610

Fuller, M. R., & Mosher, J. A. (1981). Methods of detecting and counting raptors: A review. *Studies in Avian Biology, 6*, 235:246.

Fung, T., & Siu, W.-L. (2001). A study of green space and its changes in Hong Kong using NDVI. *Geographical and Environmental Modelling, 5*. doi:10.1080/13615930120086032

Gallo, T., Fidino, M., Lehrer, E. W., & Magle, S. B. (2017). Mammal diversity and metacommunity dynamics in urban green spaces: implications for urban wildlife conservation. *Ecological Applications, 27*(8), 2330-2341. doi:10.1002/eap.1611

Gamez-Virues, S., Perovic, D. J., Gossner, M. M., Borsching, C., Bluthgen, N., de Jong, H., . . . Westphal, C. (2015). Landscape simplification filters species traits and drives biotic homogenization. *Nature Communications, 6*. doi:10.1038/ncomms9568

García-Navas, V., & Thuiller, W. (2020). Farmland bird assemblages exhibit higher functional and phylogenetic diversity than forest assemblages in France. *Journal of Biogeography, 47*(11), 2392-2404. doi: 10.1111/jbi.13950

Gates, J. E. (1997). Point Count Modifications and Breeding Bird Abundances in Central Appalachian Forests. In J. Ralph, J. R. Sauer, & S. Droege (Eds.), *Monitoring Bird Populations by Point Counts* (pp. 135-144). California: Pacific SouthWest Research Station.

González-Oreja, J. A., Bonanche-Regidor, C., Buzo-Franco, D., De la Fuente Díaz Ordaz, A. A., & Hernández-Satín, L. (2007). Caracterización ecológica de la avifauna de los parques urbanos de la ciudad de Puebla (México). *Ardeola, 54*(1), 53-67.

González-Oreja, J. A., De la Fuente Díaz Ordaz, A. A., Hernández Satín, L., Buzo Franco, D., & Bonache-Regidor, C. (2010). Evaluación de estimadores no paramétricos de la riqueza de especies. Un ejemplo con aves en áreas verdes de la ciudad de Puebla, México. *Animal Biodiversity and Conservation, 33*, 31-45.

Gonzalez-Oreja, J. A., De La Fuente-Diaz-Ordaz, A. A., Hernandez-Santin, L., Bonache-Regidor, C., & Buzo-Franco, D. (2012). Can human disturbance promote nestedness? Songbirds and noise in urban parks as a case study. *Landscape and Urban Planning, 104*(1), 9-18. doi:10.1016/j.landurbplan.2011.09.001

Graham, S. I., Kinnaird, M. F., O'Brien, T. G., Vagen, T. G., Winowiecki, L. A., Young, T. P., & Young, H. S. (2019). Effects of land-use change on community diversity and composition are highly variable among functional groups. *Ecological Applications*. doi:10.1002/eap.1973

Grajales, K. M. (2009). *Efecto de la urbanización sobre la estructura de las comunidades de aves en la Ciudad de Durango, Durango.* (Tesis de maestría), Instituto Politécnico Nacional, Victoria de Durango, México.

Grimm, N. B., Faeth, S. H., Golubiewski, N. E., Redman, C. L., Wu, J. G., Bai, X. M., & Briggs, J. M. (2008). Global change and the ecology of cities. *Science, 319*(5864), 756-760. doi:10.1126/science.1150195

Grueber, C. E., Nakagawa, S., Laws, R. J., & Jamieson, I. G. (2011). Multimodel inference in ecology and evolution: challenges and solutions. *Journal of Evolutionary Biology, 24*(4), 699-711. doi:10.1111/j.1420-9101.2010.02210.x

Han, D., Zhang, C., Wang, C., She, J. Y., Sun, Z. K., Zhao, D. X., . . . Cheng, H. (2021). Differences in Response of Butterfly Diversity and Species Composition in Urban Parks to Land Cover and Local Habitat Variables. *Forests, 12*(2). doi:10.3390/f12020140

Harvey, C. A., Medina, A., Sánchez, D. M., Vílchez, S., Hernández, B., Saenz, J. C., . . . Sinclair, F. L. (2006). Patterns of animal diversity in different forms of tree cover in agricultural landscapes. *Ecological Applications, 16*(5), 1986-1999. doi:10.1890/1051-0761(2006)016[1986:poadid]2.0.co;2

Hashim, H., Abd Latif, Z., & Adnan, N. A. (2019). Urban Vegetation Classification with NDVI Threshold Value Method with Very High Resolution (VHR) Pleiades Imagery. *International Archives of the Photogrammetry, Remote Sensing and Spatial Information Sciences, XLII-4/W16*, 237-240. doi:10.5194/isprs-archives-XLII-4-W16-237-2019

Hayes, W. M., Fisher, J. C., Pierre, M. A., Bicknell, J. E., & Davies, Z. G. (2020). Bird communities across varying landcover types in a Neotropical city. *Biotropica, 52*(1), 151-164. doi:10.1111/btp.12729

Hernández, A., J. A. Lobato y G. Contreras., (2013). Listado de avistamientos de avifauna identificada haciendo uso de hábitat en las áreas verdes del CBTIS 165 de Coatepec, Veracruz. Pp. 131. En XII Congreso para el estudio y conservación de las aves en México (CECAM). Sociedad para el Estudio y Conservación de las Aves en México A. C. San Cristóbal de las Casas.

Hernández, J. y E. Peláez, (2012). Análisis de la Avifauna del Jardín etnobotánico Francisco Peláez. Elementos, 86: 25–29.

Hernández-Cerda, M. E., Villicaña-Cruz, F. J., & Azpra-Romero, E. (2016). Clima. In C. N. p. e. C. y. U. d. l. B. C. y. S. d. M. A. (SEDEMA) (Ed.), *La biodiversidad en la Ciudad de México, Volumen I* (pp. 36-55). México: CONABIO/SEDEMA.

Hooper, D. U., Chapin III, F. S., Ewel, J. J., Hector, A., Inchausti, P., Lavorel, S., . . . Wardle, D. A. (2005). Effects of Biodiveristy on Ecosystem Functioning: A Consensus of Current Knowledge. *Ecological Monographs, 75*(1), 3-35. doi:10.1890/04-0922

Huete, A. R., Jackson, R. D., & Post, D. F. (1985). Spectral Response of a Plant Canopy with Different Soil Backgrounds. *Remote Sensing of Environment, 17*, 37-53.

Huete, A. R. (1988). A soil-adjusted vegetation index (SAVI). *Remote Sensing of Environment, 25*(3), 295-309. doi: 10.1016/0034-4257(88)90106-X

Huste, A., & Boulinier, T. (2011). Determinants of bird community composition on patches in the suburbs of Paris, France. *Biological Conservation, 144*(1), 243-252. doi:10.1016/j.biocon.2010.08.022

Hutto, R. L., Pletschet, S. M., & Hendricks, P. (1986). A Fixed-radius Point Count Method for Nonbreeding and Breeding Season Use. *The Auk, 103*(3), 593-602. doi:10.1093/auk/103.3.593

Ibáñez-Álamo, J. D., Morelli, F., Benedetti, Y., Rubio, E., Jokimäki, J., Pérez-Contreras, T., . . . Díaz, M. (2020). Biodiversity within the city: Effects of land sharing and land sparing urban development on avian diversity. *Science of The Total Environment, 707*, 135477. doi:10.1016/j.scitotenv.2019.135477

Iknayan, K. J., Tingley, M. W., Furnas, B. J., & Beissinger, S. R. (2014). Detecting diversity: emerging methods to estimate species diversity. *Trends in Ecology & Evolution, 29*(2), 97-106. doi:10.1016/j.tree.2013.10.012

Instituto Nacional de Estadística y Geografía (INEGI) (2016). Conociendo Ciudad de México. Sixth Edition. Recuperado de http://www.sideso.cdmx.gob.mx/documentos/2017/diagnostico/inegi/2016/Conociendo%20la%20Ciudad%20de%20Mexico%202016.pdf

Instituto Nacional de Estadística y Geografía (INEGI) (2017). Anuario estadístico y geográfico de la Ciudad de México 2017. INEGI, Aguascalientes, México.

Instituto Nacional de Estadística y Geografía (INEGI) (2021). Censo de Población y Vivienda 2020. Recuperado de: https://www.ingei.org.mx/programas/ccpv/2020/default.html#Resultados_generales

Jacoboski, L. I., Luza, A. L., Paulsen, R. K., Pezda, A. M., & Hartz, S. M. (2019). The effects of grassland ecosystem afforestation on avian phylogenetic diversity, taxonomic diversity and evolutionary distinctiveness. *Acta Oecologica-International Journal of Ecology, 99*. doi:10.1016/j.actao.2019.103449

Jetz, W., Thomas, G. H., Joy, J. B., Hartmann, K., & Mooers, A. O. (2012). The global diversity of birds in space and time. *Nature, 491*(7424), 444-448. doi:10.1038/nature11631

Jiménez-Moreno, F. J., & Mendoza-Cuamatzi, R. (2010). Aves urbanas en Ciudad Universitaria de la BUAP. *Elementos, 79*, 23-27.

Kembel, S.W., Cowan, P.D., Helmus, M.R., Cornwell, W.K., Morlon, H., Ackerly, D.D., Blomberg, S.P., Webb, C.O., (2010). Picante: R tools for integrating phylogenies and ecology. Bioinformatics 26:1463-1464.

Kleyer, M., Dray, S., Bello, F., Lepš, J., Pakeman, R. J., Strauss, B., . . . Lavorel, S. (2012). Assessing species and community functional responses to environmental gradients: which multivariate methods? *Journal of Vegetation Science, 23*(5), 805-821. doi:doi:10.1111/j.1654-1103.2012.01402.x

Kraft, N. J. B., Adler, P. B., Godoy, O., James, E. C., Fuller, S., & Levine, J. M. (2015). Community assembly, coexistence and the environmental filtering metaphor. *Functional Ecology, 29*(5), 592-599. doi:10.1111/1365-2435.12345

Kunc, H. P., & Schmidt, R. (2021). Species sensitivities to a global pollutant: A meta-analysis on acoustic signals in response to anthropogenic noise. *Global Change Biology, 27*(3), 675-688. doi: 10.1111/gcb.15428

La Sorte, F. A., Lepczyk, C. A., Aronson, M. F. J., Goddard, M. A., Hedblom, M., Katti, M., . . . Yang, J. (2018). The phylogenetic and functional diversity of regional breeding bird assemblages is reduced and constricted through urbanization. *Diversity and Distributions, 24*(7), 928-938. doi:10.1111/ddi.12738

La Sorte, F. A., Aronson, M. F. J., Lepczyk, C. A., & Horton, K. G. (2020). Area is the primary correlate of annual and seasonal patterns of avian species richness in urban green spaces. *Landscape and Urban Planning, 203*. doi:10.1016/j.landurbplan.2020.103892

Laliberté, E., Legendre, P., Shipley B., 2014. FD: measuring functional diversity from multiple traits, and other tools for functional ecology. R package version 1.0-12.

Lees, A. C., & Moura, N. G. (2017). Taxonomic, phylogenetic and functional diversity of an urban Amazonian avifauna. *Urban Ecosystems, 20*(5), 1019-1025. doi:10.1007/s11252-017-0661-6

Legendre, P., & Gallagher, E. D. (2001). Ecologically meaningful transformations for ordination of species data. *Oecologia, 129*(2), 271-280. doi:10.1007/s004420100716

Leston, L. F. V., & Rodewald, A. D. (2006). Are urban forests ecological traps for understory birds? An examination using Northern cardinals. *Biological Conservation, 131*(4), 566-574. doi:10.1016/j.biocon.2006.03.003

Leveau, L., Isla, F., & Bellocq, M. (2018). Predicting the seasonal dynamics of bird communities along an urban-rural gradient using NDVI. *Landscape and Urban Planning, 177*. doi:10.1016/j.landurbplan.2018.04.007

Leveau, L. M., Isla, F. I., & Isabel Bellocq, M. (2020). From town to town: Predicting the taxonomic, functional and phylogenetic diversity of birds using NDVI. *Ecological Indicators, 119*, 106703. doi:10.1016/j.ecolind.2020.106703

Leveau, L. M., & Leveau, C. M. (2016). Does urbanization affect the seasonal dynamics of bird communities in urban parks? *Urban Ecosystems, 19*(2), 631-647. doi:10.1007/s11252-016-0525-5

Leveau, L. M., & Leveau, C. M. (2020). Street design in suburban areas and its impact on bird communities: Considering different diversity facets over the year. *Urban Forestry & Urban Greening, 48*, 126578. doi: 10.1016/j.ufug.2019.126578

Lim, H. C., & Sodhi, N. S. (2004). Responses of avian guilds to urbanisation in a tropical city. *Landscape and Urban Planning, 66*(4), 199-215. doi:10.1016/s0169-2046(03)00111-7

Litteral, J., & Wu, J. G. (2012). Urban landscape matrix affects avian diversity in remnant vegetation fragments: Evidence from the Phoenix metropolitan region, USA. *Urban Ecosystems, 15*(4), 939-959. doi:10.1007/s11252-012-0245-4

Liu, J., Bai, H., Ma, H., & Feng, G. (2019). Bird diversity in Chinese urban parks was more associated with natural factors than anthropogenic factors. *Urban Forestry & Urban Greening, 43*, 126358. doi:10.1016/j.ufug.2019.06.001

Loss, S. R., Ruiz, M. O., & Brawn, J. D. (2009). Relationships between avian diversity, neighborhood age, income, and environmental characteristics of an urban landscape. *Biological Conservation, 142*(11), 2578-2585. doi:10.1016/j.biocon.2009.06.004

MacDonald, D. H., Crossman, N. D., Mahmoudi, P., Taylor, L. O., Summers, D. M., & Boxall, P. C. (2010). The value of public and private green spaces under water restrictions. *Landscape and Urban Planning, 95*(4), 192-200. doi:10.1016/j.landurbplan.2010.01.003

MacGregor-Fors, I. (2005). Listado ornitológico del Centro Universitario de Ciencias Biológicas y Agropecuarias, Universidad de Guadalajara, Jalisco, México: un espacio suburbano. *Huitzil, 6*(1), 1-6.

MacGregor-Fors, I. (2008). Relation between habitat attributes and bird richness in a western Mexico suburb. *Landscape and Urban Planning, 84*(1), 92-98. doi:10.1016/j.landurbplan.2007.06.010

MacGregor-Fors, I., Avendano-Reyes, S., Bandala, V. M., Chacon-Zapata, S., Diaz-Toribio, M., Gonzalez-Garcia, F., . . . Escobar, F. (2015). Multi-taxonomic diversity patterns in a neotropical green city: a rapid biological assessment. *Urban Ecosystems, 18*(2), 633-647. doi:10.1007/s11252-014-0410-z

MacGregor-Fors, I., Escobar, F., Rueda-Hernandez, R., Avendano-Reyes, S., Baena, M. L., Bandala, V. M., . . . Utrera-Barrillas, E. (2016). City "Green" Contributions: The Role of Urban Greenspaces as Reservoirs for Biodiversity. *Forests, 7*(7). doi:10.3390/f7070146

MacGregor-Fors, I., Morales-Pérez, L., & Schondube, J. E. (2010). Migrating to the City: Responses of Neotropical Migrant Bird Communities to Urbanization. *The Condor, 112*(4), 711-717. doi:10.1525/cond.2010.100062

MacGregor-Fors, I., & Ortega-Álvarez, R. (2011). Fading from the forest: Bird community shifts related to urban park site-specific and landscape traits. *Urban Forestry & Urban Greening, 10*(3), 239-246. doi:10.1016/j.ufug.2011.03.004

MacGregor-Fors, I., & Schondube, J. E. (2011). Gray vs. green urbanization: Relative importance of urban features for urban bird communities. *Basic and Applied Ecology, 12*(4), 372-381. doi:10.1016/j.baae.2011.04.003

Magle, S. B., Hunt, V. M., Vernon, M., & Crooks, K. R. (2012). Urban wildlife research: Past, present, and future. *Biological Conservation, 155*, 23-32. doi: 10.1016/j.biocon.2012.06.018

Magurran, A. E. (2004). *Measuring Biological Diversity*. Oxford: Blackwell Publishing.

Malagamba-Rubio, A., MacGregor-Fors, I., & Pineda-López, R. (2013). Comunidades de aves en áreas verdes de la Ciudad de Santiago de Querétaro, México. *Ornitologia Neotropical, 24*(4), 371-386.

Martensen, A. C., Pimentel, R. G., & Metzger, J. P. (2008). Relative effects of fragment size and connectivity on bird community in the Atlantic Rain Forest: Implications for conservation. *Biological Conservation, 141*(9), 2184-2192. doi: 10.1016/j.biocon.2008.06.008

Martensen, A. C., Ribeiro, M. C., Banks-Leite, C., Prado, P. I., & Metzger, J. P. (2012). Associations of Forest Cover, Fragment Area, and Connectivity with Neotropical Understory Bird Species Richness and Abundance. *Conservation Biology, 26*(6), 1100-1111. doi:10.1111/j.1523-1739.2012.01940.x

Martínez-Gardea, B. D. (2011). *Avifauna de dos áreas verdes: Parque central poniente y Club campestre en Ciudad Juárez, Chihuahua.* (Tesis de licenciatura), Universidad Autónoma de Ciudad Juárez, Ciudad Juárez, México.

Marzluff, J. M. (2017). A decadal review of urban ornithology and a prospectus for the future. *Ibis, 159*(1), 1-13. doi:10.1111/ibi.12430

Mason, N. W. H., & Mouillot, D. (2013). Functional Diversity Measures. In S. A. Levin (Ed.), *Encyclopedia of Biodiversity (Second Edition)* (pp. 597-608). Waltham: Academic Press.

Matthies, S. A., Rüter, S., Schaarschmidt, F., & Prasse, R. (2017). Determinants of species richness within and across taxonomic groups in urban green spaces. *Urban Ecosystems, 20*(4), 897-909. doi:10.1007/s11252-017-0642-9

Maya-Elizarrarás, E. (2011). Aves explotadoras de áreas verdes urbanas: un ejemplo de la zona metropolitana de Guadalajara. *El canto del Centzontle, 2*, 104-109.

Mazerolle, M., Marc, A., & Villard, M.-A. (1999). Patch characteristics and landscape context as predictors of species presence and abundance: A review. *Ecoscience, 6*, 117-124. doi:10.1080/11956860.1999.11682524

McDonald, R. I., Mansur, A. V., Ascensão, F., Colbert, M. l., Crossman, K., Elmqvist, T., . . . Ziter, C. (2020). Research gaps in knowledge of the impact of urban growth on biodiversity. *Nature Sustainability, 3*(1), 16-24. doi:10.1038/s41893-019-0436-6

McGill, B. J., Enquist, B. J., Weiher, E., & Westoby, M. (2006). Rebuilding community ecology from functional traits. *Trends in Ecology & Evolution, 21*(4), 178-185. doi:10.1016/j.tree.2006.02.002

McKinney, M. L. (2002). Urbanization, biodiversity, and conservation. *Bioscience, 52*(10), 883-890. doi:10.1641/0006-3568(2002)052[0883:ubac]2.0.co;2

Meffert, P. J., & Dziock, F. (2013). The influence of urbanisation on diversity and trait composition of birds. *Landscape Ecology, 28*(5), 943-957. doi:10.1007/s10980-013-9867-z

Meléndez-Herrada, A., Gómez de Silva, H., & Ortega-Álvarez, R. (2016). Aves. In C. N. p. e. C. y. U. d. l. B. C. y. S. d. M. A. (SEDEMA) (Ed.), *La biodiversidad en la Ciudad de México, Volumen II* (pp. 404-413). México: CONABIO/SEDEMA.

Melles, S., Glenn, S., & Martin, K. (2003). Urban bird diversity and landscape complexity: Species-environment associations along a multiscale habitat gradient. *Conservation Ecology, 7*(1).

Morelli, F., Benedetti, Y., Ibanez-Alamo, J. D., Jokimaki, J., Mand, R., Tryjanowski, P., & Moller, A. P. (2016). Evidence of evolutionary homogenization of bird communities in urban environments across Europe. *Global Ecology and Biogeography, 25*(11), 1284-1293. doi:10.1111/geb.12486

Morelli, F., Benedetti, Y., Su, T., Zhou, B., Moravec, D., Šímová, P., & Liang, W. (2017). Taxonomic diversity, functional diversity and evolutionary uniqueness in bird communities of Beijing's urban parks: Effects of land use and vegetation structure. *Urban Forestry & Urban Greening, 23*, 84-92. doi:10.1016/j.ufug.2017.03.009

Morelli, F., Mikula, P., Benedetti, Y., Bussiere, R., & Tryjanowski, P. (2018). Cemeteries support avian diversity likewise urban parks in European cities: Assessing taxonomic, evolutionary and functional diversity. *Urban Forestry & Urban Greening, 36*, 90-99. doi:10.1016/j.ufug.2018.10.011

Moreno-Contreras, I., Gómez de Silva, H., Torres-Vivanco, A., Villalpando-Navarrete, N., & Botello, A. (2015). Avifauna of Juárez Municipality, Chihuahua, México. *Western Birds, 46*(3), 190-2015.

Murgui, E. (2010). Seasonality and nestedness of bird communities in urban parks in Valencia, Spain. *Ecography, 33*(5), 979-984. doi:10.1111/j.1600-0587.2010.05816.x

Murgui, E., & Hedblom, M. (2017). *Ecology and Conservation of Birds in Urban Environments*: Springer.

Naeem, S., Prager, C., Weeks, B., Varga, A., Flynn, D. F. B., Griffin, K., . . . Schuster, W. (2016). Biodiversity as a multidimensional construct: a review, framework and case study of herbivory's impact on plant biodiversity. *Proceedings of the Royal Society B-Biological Sciences, 283*(1844). doi:10.1098/rspb.2015.3005

Nagendra, H. (2002). Opposite trends in response for the Shannon and Simpson indices of landscape diversity. *Applied Geography, 22*(2), 175-186. doi:10.1016/s0143-6228(02)00002-4

Nava-Díaz, R. (2016). Diversidad de aves en áreas verdes de zonas urbanas: una revisión para México. In A. Ramírez-Bautista & R. Pineda-López (Eds.), *Fauna Nativa en Ambientes Antropizados* (pp. 51-63). Querétaro, México: CONACYT-UAQ.

Nava-Díaz, R., Pineda-López, R., & Dorantes-Euan, A. (2020). Drivers of Functional Composition of Bird Assemblages in Green Spaces of a Neotropical City: A Case Study From Merida, Mexico. *Tropical Conservation Science, 13*, 1940082920923896. doi:10.1177/1940082920923896

Navarro-Sigüenza, A. G., Rebón-Gallardo, M. F., Gordillo-Martínez, A., Peterson, T., Berlanga-García, H., & Sánchez-González, L. A. (2014). Biodiversidad de aves en México. *Revista Mexicana de Biodiversidad, 84*, 476-495.

Newbold, T., Hudson, L. N., Hill, S. L. L., Contu, S., Lysenko, I., Senior, R. A., . . . Purvis, A. (2015). Global effects of land use on local terrestrial biodiversity. *Nature, 520*(7545), 45-50. doi:10.1038/nature14324

Nielsen, A. B., van den Bosch, M., Maruthaveeran, S., & van den Bosch, C. K. (2014). Species richness in urban parks and its drivers: A review of empirical evidence. *Urban Ecosystems, 17*(1), 305-327. doi:10.1007/s11252-013-0316-1

Niemela, J. (2011). *Urban Ecology*. Great Britain: Oxford University Press.

Odindi, J., & Mhangara, P. (2012). Green Spaces Trends in the City of Port Elizabeth from 1990 to 2000 using Remote Sensing. *International Journal of Environmental Research, 6*, 653-662.

Ofori, B. Y., Garshong, R. A., Gbogbo, F., Owusu, E. H., & Attuquayefio, D. K. (2018). Urban green area provides refuge for native small mammal biodiversity in a rapidly expanding city in Ghana. *Environmental Monitoring and Assessment, 190*(8). doi:10.1007/s10661-018-6858-1

Oksanen, J., Blanchet, F. G., Friendly, M., Kindt, R., Legendre, P., McGlinn, D., . . . Wagner, H. (2017). vegan: Community Ecology Package. R package version 2.4-4: https://CRAN.R-project.org/package=vegan.

Organización de las Naciones Unidas (ONU) (2018). World Urbanization Prospects: The 2018 Revision. Recuperado de https://population.un.org/wup/Country-Profiles/

Ortega-Álvarez, R., & MacGregor-Fors, I. (2009). Living in the big city: Effects of urban land-use on bird community structure, diversity, and composition. *Landscape and Urban Planning, 90*(3-4), 189-195. doi:10.1016/j.landurbplan.2008.11.003

Ortega-Álvarez, R., & MacGregor-Fors, I. (2010). What matters most? Relative effect of urban habitat traits and hazards on urban park birds. *Ornitologia Neotropical, 21*, 519-533.

Ortega-Álvarez, R., & MacGregor-Fors, I. (2011). Dusting-off the file: A review of knowledge on urban ornithology in Latin America. *Landscape and Urban Planning, 101*, 1-10.

Pablo-López, R. E., & Díaz-Porras, D. F. (2011). Los campus universitarios como refugios de aves: el caso de la Universidad Autónoma “Benito Juárez” de Oaxaca (UABJO), Oaxaca, México. *El canto del Centzontle, 2*, 48-63.

Parra, L. G., M. Reyes, I. Núñez y A. Tovar., (2010). Monitoreo de la ornitofauna de dos zonas (norte y sur) del Distrito Federal. Pp. 60. En XVI Simposio Nacional de Ornitología. Sociedad Mexicana de Ornitología, A. C. Ciudad de México.

Parris, K. M. (2016). *Ecology of Urban Environments*. India: Wiley Blackwell.

Pater, L. L., Grubb, T. G., & Delaney, D. K. (2009). Recommendations for Improved Assessment of Noise Impacts on Wildlife. *Journal of Wildlife Management, 73*(5), 788-795. doi:10.2193/2006-235

Pavoine, S., & Bonsall, M. B. (2011). Measuring biodiversity to explain community assembly: a unified approach. *Biological Reviews, 86*(4), 792-812. doi:10.1111/j.1469-185X.2010.00171.x

Peraza-Contreras, G. C. (2011). *Vegetación nativa para el diseño de espacios públicos en la ciudad de Mérida* (major's thesis). Universidad Nacional Autónoma de México, México City, México.

Petchey, O. L., Evans, K. L., Fishburn, I. S., & Gaston, K. J. (2007). Low functional diversity and no redundancy in British avian assemblages. *Journal of Animal Ecology, 76*(5), 977-985. doi:10.1111/j.1365-2656.2007.01271.x

Planillo, A., Kramer-Schadt, S., Buchholz, S., Gras, P., von der Lippe, M., & Radchuk, V. (2021). Arthropod abundance modulates bird community responses to urbanization. *Diversity and Distributions, 27*(1), 34-49. doi:10.1111/ddi.13169

Purevdorj, T. S., Tateishi, R., Ishiyama, T., & Honda, Y. (1998). Relationships between percent vegetation cover and vegetation indices. *International Journal of Remote Sensing, 19*(18), 3519-3535. doi:10.1080/014311698213795

QGIS Development Team, (2002). QGIS Geographic Information System. Open Source Foundation Project. http://qgis.osgeo.org

R Core Team, (2018). R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. URL https://www.R-project.org/

Ralph, C. J., Geupel, G. R., Pyle, P., Martin, T. E., DeSante, D. F., & Milá, B. (1996). *Manual de métodos de campo para el monitoreo de aves terrestres*. Albany: U.S. Department of Agriculture.

Ramírez-Albores, J. E. (2008). Comunidad de aves de la Facultad de Estudios Superiores Zaragoza campus II, UNAM, Ciudad de México. *Huitzil, 9*(2), 12-19.

Resendiz, N. H., J. Cipriano y R. Valencia., (2013). Avifauna de áreas verdes urbanas de la ciudad de Huejutla de Reyes, Hidalgo, México. Pp. 60-61. En XII Congreso para el estudio y conservación de las aves en México (CECAM). Sociedad para el Estudio y Conservación de las Aves en México A. C. San Cristóbal de las Casas.

Reyna-Bustos, O. F., Cayo-Cervantes, L. A., & González-Rodríguez, A. N.-. (2013). Los Parques Urbanos de Guadalajara y su Importancia para las Aves. In A. I. Ramírez-Quintana, M. Anaya-Corona, F. D. Rentería-Rodríguez, J. G. Ruvalcaba-Salazar, & M. A. González-Villa (Eds.), *Propuestas para la gestión de los Parques en México 2012* (pp. 77-90). Jalisco, México: Orgánica.

Rico-Silva, J. F., Cruz-Trujillo, E. J., & Colorado Z, G. J. (2020). Correction to: Influence of environmental factors on bird diversity in greenspaces in an Amazonian city. *Urban Ecosystems*. doi:10.1007/s11252-020-01051-w

Roberts, H. P., & King, D. I. (2017). Area requirements and landscape-level factors influencing shrubland birds. *Journal of Wildlife Management, 81*(7), 1298-1307. doi:10.1002/jwmg.21286

Rodríguez Ruiz, E. R. (2007). Estudio preliminar de la ornitofauna en el Instituto Tecnológico de Cd. Victoria Tamaulipas, México. *TecnoINTELECTO, 4*(2), 63-66.

Rodríguez, T., C. Vázquez y L. Montañes., (2010). Estudio de las aves urbanas de la segunda sección del Bosque de Chapultepec, México, D.F. Pp. 46. En XVI Simposio Nacional de Ornitología. Sociedad Mexicana de Ornitología, A. C. Ciudad de México.

Rosa, S., Palmeirim, J. M., & Moreira, F. (2003). Factors affecting waterbird abundance and species richness in an increasingly urbanized area of the Tagus estuary in Portugal. *Waterbirds, 26*(2), 226-232. doi:10.1675/1524-4695(2003)026[0226:fawaas]2.0.co;2

Rubolini, D., Liker, A., Garamszegi, L. Z., Møller, A. P., & Saino, N. (2015). Using the BirdTree.org website to obtain robust phylogenies for avian comparative studies: A primer. *Current Zoology, 61*(6), 959-965. doi:10.1093/czoolo/61.6.959

Ruelas Inzunza, E., & Aguilar Rodríguez, S. H. (2010). La avifauna urbana del parque Ecológico Macuitépetl en Xalapa, Veracruz, México. *Ornitologia Neotropical, 21*(1), 87-103.

Rush, S. A., Romito, T., & Robison, T. L. (2014). Avian diversity in a suburban park system: current conditions and strategies for dealing with anticipated change. *Urban Ecosystems, 17*(1), 45-60. doi:10.1007/s11252-013-0310-7

Sacco, A. G., Rui, A. M., Bergmann, F. B., Mueller, S. C., & Hartz, S. M. (2015). Reduction in taxonomic and functional bird diversity in an urban area in Southern Brazil. *Iheringia Serie Zoologia, 105*(3), 276-287. doi:10.1590/1678-476620151053276287

San José, M., Garmendia, A., & Cano-Santana, Z. (2010). Monitoreo de aves en dos zonas de restauración ecológica y una de referencia, en la Reserva Ecológica del Pedregal de San Ángel, D. F., México. *El canto del Centzontle, 1*(2), 148-164.

Scherer, A., Scherer, S., Bugoni, L., Mohr, L., Efe, M., & Hartz, S. M. (2005). Estrutura trófica da Avifauna em oito parques da cidade de Porto Alegre, Rio Grande do Sul, Brasil. *Ornithologia, 1*(1), 25-32.

Schmidt, J. H., McIntyre, C. L., & MacCluskie, M. C. (2013). Accounting for incomplete detection: What are we estimating and how might it affect long-term passerine monitoring programs?. *Biological Conservation, 160*, 130-139. doi:10.1016/j.biocon.2013.01.007

Schütz, C., & Schulze, C. H. (2015). Functional diversity of urban bird communities: effects of landscape composition, green space area and vegetation cover. *Ecology and Evolution, 5*(22), 5230-5239. doi:10.1002/ece3.1778

Secretaría de Medio Ambiente y Recursos Naturales (SERMANAT). 2010. NORMA Oficial Mexicana NOM-059-SERMANAT-2010, Protección ambiental-Especies nativas de México de flora y fauna silvestres-Categorías de riesgo y especificaciones para su inclusión, exclusión o cambio-Lista de especies en riesgo. Diario Oficial de la Federación.

Seto, K. C., Fragkias, M., Güneralp, B., & Reilly, M. K. (2011). A Meta-Analysis of Global Urban Land Expansion. *Plos One, 6*(8), e23777. doi:10.1371/journal.pone.0023777

Seto, K. C., Güneralp, B., & Hutyra, L. R. (2012). Global forecasts of urban expansion to 2030 and direct impacts on biodiversity and carbon pools. *Proceedings of the National Academy of Sciences, 109*(40), 16083-16088. doi:10.1073/pnas.1211658109

Shanahan, D. F., Miller, C., Possingham, H. P., & Fuller, R. A. (2011). The influence of patch area and connectivity on avian communities in urban revegetation. *Biological Conservation, 144*(2), 722-729. doi:10.1016/j.biocon.2010.10.014

Shwartz, A., Muratet, A., Simon, L., & Julliard, R. (2013). Local and management variables outweigh landscape effects in enhancing the diversity of different taxa in a big metropolis. *Biological Conservation, 157*, 285-292. doi:10.1016/j.biocon.2012.09.009

Silva, C. P., Sepulveda, R. D., & Barbosa, O. (2016). Nonrandom filtering effect on birds: species and guilds response to urbanization. *Ecology and Evolution, 6*(11), 3711-3720. doi:10.1002/ece3.2144

Sims, V., Evans, K. L., Newson, S. E., Tratalos, J. A., & Gaston, K. J. (2008). Avian assemblage structure and domestic cat densities in urban environments. *Diversity and Distributions, 14*(2), 387-399. doi:10.1111/j.1472-4642.2007.00444.x

Sobrino, J. A., Jiménez-Muñoz, J. C., & Paolini, L. (2004). Land surface temperature retrieval from LANDSAT TM 5. *Remote Sensing of Environment, 90*(4), 434-440. doi:10.1016/j.rse.2004.02.003

Sol, D., Bartomeus, I., Gonzalez-Lagos, C., & Pavoine, S. (2017). Urbanisation and the loss of phylogenetic diversity in birds. *Ecology Letters, 20*(6), 721-729. doi:10.1111/ele.12769

Song, M.-K., Lin, F.-C., Ward, S. E., & Fine, J. P. (2013). Composite variables: when and how. *Nursing research, 62*(1), 45-49. doi:10.1097/NNR.0b013e3182741948

Stofberg, M., Cunningham, S. J., Sumasgutner, P., & Amar, A. (2019). Juggling a "junk-food" diet: responses of an urban bird to fluctuating anthropogenic-food availability. *Urban Ecosystems, 22*. doi:10.1007/s11252-019-00885-3

Suarez-Rubio, M., & Thomlinson, J. R. (2009). Landscape and patch-level factors influence bird communities in an urbanized tropical island. *Biological Conservation, 142*(7), 1311-1321. doi:10.1016/j.biocon.2008.12.035

Supp, S. R., & Ernest, S. K. M. (2014). Species-level and community-level responses to disturbance: a cross-community analysis. *Ecology, 95*(7), 1717-1723. doi:10.1890/13-2250.1

Swenson, N. G. (2014). Functional Diversity. In N. G. Swenson (Ed.), *Functional and Phylogenetic Diversity in R* (pp. 143-186). New York: Springer.

Symonds, M. R. E., & Moussalli, A. (2011). A brief guide to model selection, multimodel inference and model averaging in behavioural ecology using Akaike's information criterion. *Behavioral Ecology and Sociobiology, 65*(1), 13-21. doi:10.1007/s00265-010-1037-6

ter Braak, C. J. F., Cormont, A., & Dray, S. (2012). Improved testing of species traits–environment relationships in the fourth-corner problem. *Ecology, 93*(7), 1525-1526. doi:10.1890/12-0126.1

Thioulouse, J., Dray, S., Dufour, A.-B., Siberchicot, A., Jombart, T., & Pavoine, S. (2018). *Multivariate Analysis of Ecological Data with ade4*. New York, USA: Springer.

Tobias, J. A., & Pigot, A. L. (2019). Integrating behaviour and ecology into global biodiversity conservation strategies. *Philosophical Transactions of the Royal Society B: Biological Sciences, 374*(1781), 20190012. doi:10.1098/rstb.2019.0012

Torrescano-Valle, N., & Folan, W. J. (2015). Physical Settings, Environmental History with an Outlook on Global Change. In G. A. Islebe, S. Calmé, J. L. León-Cortés, & B. Schmook (Eds.), *Biodiversity and Conservation of the Yucatán Peninsula* (pp. 9-37): Springer.

Tryjanowski, P., Morelli, F., Mikula, P., Krištín, A., Indykiewicz, P., Grzywaczewski, G., . . . Jerzak, L. (2017). Bird diversity in urban green space: A large-scale analysis of differences between parks and cemeteries in Central Europe. *Urban Forestry & Urban Greening, 27*, 264-271. doi:10.1016/j.ufug.2017.08.014

Tryjanowski, P., Sparks, T., Biaduń, W., Brauze, T., Hetmański, T., Martyka, R., . . . Wysocki, D. (2015). Winter Bird Assemblages in Rural and Urban Environments: A National Survey. *Plos One, 10*, e0130299. doi:10.1371/journal.pone.0130299

Useni Sikuzani, Y., Sambiéni Kouagou, R., Maréchal, J., Ilunga wa Ilunga, E., Malaisse, F., Bogaert, J., & Munyemba Kankumbi, F. (2018). Changes in the Spatial Pattern and Ecological Functionalities of Green Spaces in Lubumbashi (the Democratic Republic of Congo) in Relation With the Degree of Urbanization. *Tropical Conservation Science, 11*, 1940082918771325. doi:10.1177/1940082918771325

Varona-Graniel, D. E. (2001). *Avifauna de áreas verdes urbanas del norte de la Ciudad de México.* (Tesis de maestría), Universidad Nacional Autónoma de México, Distrito Federal, México.

Villafranco-Castro, J. A. (2000). *Avifauna del Parque Tezozomoc, Azcapotzalco.* (Tesis de licenciatura), Universidad Nacional Autónoma de México, Distrito Federal, México.

Villarroya-Villalba, L., Casanelles-Abella, J., Moretti, M., Pinho, P., Samson, R., Van Mensel, A., . . . Obrist, M. K. (2021). Response of bats and nocturnal insects to urban green areas in Europe. *Basic and Applied Ecology, 51*, 59-70. doi:10.1016/j.baae.2021.01.006

Villéger, S., Mason, N. W. H., & Mouillot, D. (2008). New Multidimensional Functional Diversity Indices for a Multifaceted Framework in Functional Ecology. *Ecology, 89*(8), 2290-2301. doi:10.1890/07-1206.1

Voskamp, A., Baker, D. J., Stephens, P. A., Valdes, P. J., & Willis, S. G. (2017). Global patterns in the divergence between phylogenetic diversity and species richness in terrestrial birds. *Journal of Biogeography, 44*(4), 709-721. doi:10.1111/jbi.12916

Webb, C. T., Hoeting, J. A., Ames, G. M., Pyne, M. I., & Poff, N. L. (2010). A structured and dynamic framework to advance traits-based theory and prediction in ecology. *Ecology Letters, 13*(3), 267-283. doi:10.1111/j.1461-0248.2010.01444.x

Weiher, E. (2011). A primer of trait and functional diversity. In A. E. Magurran (Ed.), *Biological Diversity* (pp. 175-193). New York: Oxford University Press.

White, J. G., Antos, M. J., Fitzsimons, J. A., & Palmer, G. C. (2005). Non-uniform bird assemblages in urban environments: the influence of streetscape vegetation. *Landscape and Urban Planning, 71*(2-4), 123-135. doi:10.1016/j.landurbplan.2004.02.006

Wilman, H., Belmaker, J., Simpson, J., de la Rosa, C., Rivadeneira, M. M., & Jetz, W. (2014). EltonTraits 1.0: Species-level foraging attributes of the world's birds and mammals. *Ecology, 95*(7), 2027-2027. doi:10.1890/13-1917.1

Wolff, P. J., Degregorio, B. A., Rodriguez-Cruz, V., Mulero-Oliveras, E., & Sperry, J. H. (2018). Bird Community Assemblage and Distribution in a Tropical, Urban Ecosystem of Puerto Rico. *Tropical Conservation Science, 11*, 1-10. doi:10.1177/1940082918754777

Wong, M. M. F., Fung, J. C. H., & Yeung, P. P. S. (2019). High-resolution calculation of the urban vegetation fraction in the Pearl River Delta from the Sentinel-2 NDVI for urban climate model parameterization. *Geoscience Letters, 6*(1), 2. doi:10.1186/s40562-019-0132-4

Yang, X., Tan, X., Chen, C., & Wang, Y. (2020). The influence of urban park characteristics on bird diversity in Nanjing, China. *Avian Research, 11*(1), 45. doi:10.1186/s40657-020-00234-5

Young, H. S., McCauley, D. J., Galetti, M., & Dirzo, R. (2016). Patterns, Causes, and Consequences of Anthropocene Defaunation. *Annual Review of Ecology, Evolution, and Systematics, 47*(1), 333-358. doi:10.1146/annurev-ecolsys-112414-054142

Zar, J. H. (2014). *Biostatistical Analysis*. United States of America: Pearson.

Zungu, M. M., Maseko, M. S. T., Kalle, R., Ramesh, T., & Downs, C. T. (2020). Effects of landscape context on mammal richness in the urban forest mosaic of EThekwini Municipality, Durban, South Africa. *Global Ecology and Conservation, 21*, e00878. doi:10.1016/j.gecco.2019.e00878

Zuur, A. F., Ieno, E. N., Walker, N. J., Saveliev, A. A., & Smith, G. M. (2009). *Mixed Effects Models and Extensions in Ecology with R*: Springer.

APÉNDICES

Capítulo 2. Appendix 1. Species recorded in sampled green spaces.

Order	**Family**	**Scientific name**	**English common name**	**Code**	**gs 1**	**gs 2**	**gs 3**	**gs 4**	**gs 5**	**gs 6**	**gs 7**	**gs 8**	**gs 9**	**gs 10**	**gs 11**	**g 1**
Anseriformes	Anatidae	*Spatula discors*	Blue-winged Teal	Spadis												X
Galliformes	Cracidae	*Ortalis vetula*	Plain Chachalaca	Ortvet												
Podicipediformes	Podicipedidae	*Tachybaptus dominicus*	Least Grebe	Tacdom												X
Columbiformes	Columbidae	*Columba livia*	Rock Pigeon	Colliv	X					X						
		Streptopelia decaocto	Eurasian Collared-Dove	Strdec	X					X			X	X	X	
		Columbina talpacoti	Ruddy Ground Dove	Coltal	X	X	X			X			X	X		X
		Leptotila verreauxi	White-tipped Dove	Lepver			X									
		Zenaida asiatica	White-winged Dove	Zenasi	X	X	X	X	X	X	X		X	X	X	X

Cuculiformes	Cuculidae	*Crotophaga sulcirostris*	Grooved-billed Ani	Crosul						X		X		X
		Geococcyx velox	Lesser Roadrunner	Geovel										
		Piaya cayana	Squirrel Cuckoo	Piacay										
Apodiformes	Trochilidae	*Chlorostilbon canivetii*	Canivet's Emerald	Chlcan						X				
		Amazilia rutila	Cinnamon Hummingbird	Amarut	X	X	X		X	X	X	X	X	X	X
Gruiformes	Rallidae	*Gallinula galeata*	Common Gallinule	Galgal											X
		Fulica americana	American Coot	Fulame	X										X
Charadriiformes	Recurvirostridae	*Himantopus mexicanus*	Black-necked Stilt	Himmex											X
	Jacanidae	*Jacana spinosa*	Northern Jacana	Jacspi	X								X		X
	Scolopacidae	*Calidris himantopus*	Stilt Sandpiper	Calhim											X
Suliformes	Anhingidae	*Anhinga anhinga*	Anhinga	Anhanh	X								X		
Pelecaniformes	Ardeidae	*Ardea herodias*	Great Blue Heron	Ardher									X		
		Ardea alba	Great Egret	Ardalb	X								X		
		Egretta thula	Snowy Egret	Egrthu	X										

		Egretta caerulea	Little Blue Heron	Egrcae												X
		Butorides virescens	Green Heron	Butvir		X										X
		Nyctanassa violacea	Yellow-crowned Night-heron	Nycvio		X										
Cathartiformes	Cathartidae	*Coragyps atratus*	Black Vulture	Coratr		X	X									
Accipitriformes	Accipitridae	*Rostrhamus sociabilis*	Snail Kite	Rossoc		X										
		Rupornis magnirostris	Roadside Hawk	Rupmag		X										X
Coraciiformes	Momotidae	*Momotus lessonii*	Lesson's Momot	Momles			X									X
		Eumomota superciliosa	Turquoise-browed Momot	Eumsup			X				X	X				X
	Alcedinidae	*Megaceryle alcyon*	Belted Kingfisher	Megalc		X								X		
Piciformes	Picidae	*Melanerpes pygmaeus*	Yucatan Woodpecker	Melpyg	X		X			X						X
		Melanerpes aurifrons	Golden-fronted Woodpecker	Melaur	X	X	X	X	X	X	X	X	X	X		X
		Sphyrapicus varius	Yellow-bellied Sapsucker	Sphvar										X		
		Dryobates scalaris	Ladder-backed Woodpecker	Drysca										X		

Psittaciformes	Psittacidae	*Eupsittula nana*	Olive-throated Parakeet	Eupnan												
		Amazona autumnalis	Red-lored Parrot	Amaaut									X			
Passeriformes	Tityridae	*Tityra semifasciata*	Masked Tityra	Titsem				X								
	Tyrannidae	*Camptostoma imberbe*	Northern Beardless-Tyrannulet	Camimb								X				
		Myiarchus tuberculifer	Dusky-capped Flycatcher	Myitub								X				
		Myiarchus crinitus	Great Crested Flycatcher	Myicri								X				
		Pitangus sulphuratus	Great Kiskadee	Pitsul		X	X			X	X	X		X		X
		Myiozetetes similis	Social Flycatcher	Myisim	X	X	X			X	X	X	X	X	X	X
		Tyrannus couchii	Couch's Kingbird	Tyrcou		X	X						X			X
		Contopus cinereus	Tropical Pewee	Concin												
		Empidonax minimus	Least Flycathcer	Empmin											X	X
		Pyrocephalus rubinus	Vermillion Flycatcher	Pyrrub												X

Vireonidae	*Cyclarhis gujanensis*	Rufous-browed Peppershrike	Cycguj			X					X	X	X		X
	Vireo griseus	White-eyed Vireo	Virgri		X	X		X			X				X
	Vireo pallens	Mangrove Vireo	Virpal										X		
Corvidae	*Cyanocorax yncas*	Green Jay	Cyaync			X				X	X				
	Cyanocorax yucatanicus	Yucatan Jay	Cyayuc			X					X				X
Troglodytidae	*Troglodytes aedon*	House Wren	Troaed			X									
	Thryothorus ludovicianus	Carolina Wren	Thrlud												
	Uropsila leucogastra	White-bellied Wren	Uroleu								X				
Polioptilidae	*Polioptila caerulea*	Blue-gray Gnatcatcher	Polcae	X	X	X	X	X	X		X	X	X	X	X
Turdidae	*Turdus grayi*	Clay-colored Thrush	Turgra	X	X	X	X	X	X	X	X		X		X
Mimidae	*Dumetella carolinensis*	Gray Catbird	Dumcar												X
	Mimus gilvus	Tropical Mockingbird	Mimgil	X	X	X	X	X	X	X		X	X	X	X
Fringillidae	*Euphonia affinis*	Scrub Euphonia	Eupaff								X				

	Spinus psaltria	Lesser Goldfinch	Spipsa									X			
Passerellidae	*Arremonops rufivirgatus*	Olive Sparrow	Arrruf												
Icteridae	*Icterus spurius*	Orchard Oriole	Ictspu		X	X		X				X			
	Icterus cucullatus	Hooded Oriole	Ictcuc		X	X	X	X		X		X			X
	Icterus auratus	Orange Oriole	Ictaur			X									X
	Icterus gularis	Altamira Oriole	Ictgul		X	X						X		X	
	Molothrus aeneus	Bronzed Cowbird	Molaen											X	
	Dives dives	Melodious Blackbird	Divdiv			X	X		X	X		X	X		X
	Quiscalus mexicanus	Great-tailed Grackle	Quimex	X	X	X	X	X	X	X		X	X	X	X
Parulidae	*Seiurus aurocapilla*	Ovenbird	Seiaur												X
	Parkesia noveboracensis	Northern Waterthrush	Parnov												X
	Mniotilta varia	Black and White Warbler	Mnivar												
	Geothlypis trichas	Common Yellowthroat	Geotri												X
	Setophaga citrina	Hooded Warbler	Setcit												
	Setophaga americana	Northern Parula	Setame		X	X					X				

	Setophaga magnolia	Magnolia Warbler	Setmag			X									
	Setophaga petechia	Yellow Warbler	Setpet		X										X
	Setophaga coronata	Yellow-rumped Warbler	Setcor												X
	Setophaga dominica	Yellow-throated Warbler	Setdom	X	X	X		X							X
	Setophaga virens	Black-throated Warbler	Setvir			X									
Cardinalidae	*Piranga rubra*	Summer Tanager	Pirrub	X			X	X	X	X			X		
	Passerina caerulea	Blue Grosbeak	Pascae												X
	Passerina cyanea	Indigo Bunting	Pascya			X									X
	Passerina ciris	Painted Bunting	Pascir												X
Thraupidae	*Thraupis episcopus*	Blue-gray Tanager	Threpi		X										
	Tiaris olivaceus	Yellow-faced Grassquit	Tiaoli												
	Saltator coerulescens	Grayish Saltator	Salcoe		X	X	X	X	X	X		X	X		

Capítulo 3. Appendix 1. Species incidence and total abundance in sampled green spaces across the seasons. W: winter, DB: dry breeding, WB: wet breeding.

				Incidence			Abundance		
Order	Family	Species	Common Name	W	DB	WB	W	DB	WB
Columbiformes	Columbidae	*Columba livia*	Rock Pigeon	11	12	10	95	68	45
		Columbina inca	Inca Dove	21	23	23	151	111	109
Apodiformes	Trochilidae	*Eugenes fulgens*	Rivoli's Hummingbird	1	0	0	1	0	0
		Lampornis clemenciae	Blue-throated Mountain-gem	1	0	0	2	0	0
		Cynanthus latirostris	Broad-billed Hummingbird	3	3	1	3	3	1
		Basilinna leucotis	White-eared Hummingbird	5	6	2	15	11	3
		Saucerottia beryllina	Berylline Hummingbird	7	16	12	13	53	24
Accipitriformes	Accipitridae	*Accipiter cooperii*	Cooper's Hawk	0	0	1	0	0	1
		Parabuteo unicinctus	Harris's Hawk	1	1	0	1	2	0
Piciformes	Picidae	*Melanerpes formicivorus*	Acorn Woodpecker	1	1	1	5	4	2
		Sphyrapicus varius	Yellow-bellied Sapsucker	1	0	0	1	0	0
		Dryobates scalaris	Ladder-backed Woodpecker	4	2	2	5	3	4
Falconiformes	Falconidae	*Falco sparverius*	American Kestrel	1	0	0	1	0	0
Psittaciformes	Psittacidae	*Myiopsitta monachus*	Monk Parakeet	1	0	0	1	0	0
Passeriformes	Tyrannidae	*Tyrannus vociferans*	Cassin's Kingbird	0	1	0	0	1	0
		Mitrephanes phaeocercus	Tufted Flycatcher	4	2	2	7	4	4
		Contopus pertinax	Greater Pewee	2	0	3	5	0	3
		Empidonax minimus	Least Flycatcher	2	1	0	3	1	0
		Empidonax hammondii	Hammond's Flycatcher	8	0	0	13	0	0
		Pyrocephalus rubinus	Vermilion Flycatcher	1	4	2	2	10	4
	Vireonidae	*Vireo plumbeus*	Plumbeous Vireo	2	0	0	2	0	0
	Corvidae	*Cyanocorax yncas*	Green Jay	0	2	1	0	3	2

	Cyanocitta stelleri	Steller's Jay	2	1	1	6	2	3
	Aphelocoma ultramarina	Transvolcanic Jay	0	0	1	0	0	3
Paridae	*Poecile sclateri*	Mexican Chickadee	3	3	4	7	8	7
Aegithalidae	*Psaltriparus minimus*	Bushtit	8	16	14	33	86	125
Sittidae	*Sitta carolinensis*	White-breasted Nuthatch	2	1	3	4	1	5
Certhiidae	*Certhia americana*	Brown Creeper	2	2	2	4	3	3
Troglodytidae	*Catherpes mexicanus*	Canyon Wren	0	2	1	0	5	1
	Troglodytes aedon	House Wren	1	1	0	2	1	0
	Thryomanes bewickii	Bewick's Wren	27	23	20	42	81	59
Polioptilidae	*Polioptila caerulea*	Blue-gray Gnatcatcher	14	0	2	26	0	3
Cinclidae	*Cinclus mexicanus*	American Dipper	0	1	0	0	2	0
Regulidae	*Regulus satrapa*	Golden-crowned Kinglet	2	0	2	6	0	9
	Regulus calendula	Ruby-crowned Kinglet	27	0	3	90	0	4
Turdidae	*Myadestes occidentalis*	Brown-backed Solitaire	3	3	3	11	16	5
	Catharus aurantiirostris	Orange-billed Nightingale-Thrush	0	2	0	0	2	0
	Catharus occidentalis	Russet Nightingale-Thrush	2	0	0	4	0	0
	Catharus guttatus	Hermit Thrush	1	0	0	1	0	0
	Turdus assimilis	White-throated Thrush	3	3	1	10	9	2
	Turdus rufopalliatus	Rufous-backed Robin	11	30	16	17	142	36
	Turdus migratorius	American Robin	9	20	8	35	87	18
Mimidae	*Melanotis caerulescens*	Blue Mockingbird	0	1	0	0	2	0
	Toxostoma curvirostre	Curve-billed Thrasher	2	5	3	3	12	6
Ptiliogonatidae	*Ptiliogonys cinereus*	Gray Silky-flycatcher	4	10	4	23	17	10
Peucedramidae	*Peucedramus taeniatus*	Olive Warbler	3	1	3	3	1	3

Passeridae	*Passer domesticus*	House Sparrow	20	26	18	126	246	123
Fringillidae	*Euphonia elegantissima*	Elegant Euphonia	0	1	1	0	1	7
	Coccothraustes abeillei	Hooded Grosbeak	0	1	0	0	3	0
	Haemorhous mexicanus	House Finch	25	25	22	158	116	80
	Spinus pinus	Pine Siskin	0	2	0	0	3	0
	Spinus psaltria	Lesser Goldfinch	3	3	8	8	5	23
Passerellidae	*Spizella passerina*	Chipping Sparrow	0	1	0	0	1	0
	Arremon virenticeps	Green-striped Brushfinch	2	1	1	4	1	1
	Junco phaeonotus	Yellow-eyed Junco	3	3	3	27	43	47
	Oriturus superciliosus	Striped Sparrow	1	1	0	2	6	0
	Melospiza melodia	Song Sparrow	2	5	6	2	20	9
	Melospiza lincolnii	Lincoln's Sparrow	1	0	0	6	0	0
	Melozone fusca	Canyon Towhee	10	17	12	19	42	32
	Aimophila ruficeps	Rufous-crowned Sparrow	0	0	1	0	0	1
	Pipilo maculatus	Spotted Towhee	0	1	0	0	2	0
	Atlapetes pileatus	Rufous-capped Brushfinch	0	3	2	0	10	7
Icteridae	*Icterus wagleri*	Black-vented Oriole	0	1	0	0	2	0
	Icterus spurius	Orchard Oriole	0	1	1	0	5	6
	Icterus bullockii	Bullock's Oriole	4	1	0	5	3	0
	Icterus parisorum	Scott's Oriole	2	0	0	2	0	0
	Molothrus aeneus	Bronzed Cowbird	0	15	1	0	42	2
	Quiscalus mexicanus	Great-tailed Grackle	16	15	14	107	70	127
Parulidae	*Seiurus aurocapilla*	Ovenbird	1	0	0	1	0	0
	Mniotilta varia	Black-and-white Warbler	5	1	4	5	1	4
	Oreothlypis superciliosa	Crescent-chested Warbler	1	3	4	4	17	8
	Leiothlypis celata	Orange-crowned Warbler	2	0	0	3	0	0
	Leiothlypis ruficapilla	Nashville Warbler	12	0	0	20	0	0

	Setophaga petechia	Yellow Warbler	1	0	0	1	0	0
	Setophaga coronata	Yellow-rumped Warbler	29	1	0	156	1	0
	Setophaga nigrescens	Black-throated Gray Warbler	1	0	2	1	0	2
	Setophaga townsendi	Townsend's Warbler	4	0	1	9	0	5
	Setophaga occidentalis	Hermit Warbler	0	0	2	0	0	3
	Setophaga virens	Black-throated Green Warbler	2	0	4	4	0	6
	Basileuterus rufifrons	Rufous-capped Warbler	2	1	1	5	2	3
	Basileuterus belli	Golden-browed Warbler	2	1	0	4	3	0
	Cardellina pusilla	Wilson's Warbler	18	1	1	36	1	3
	Cardellina rubra	Red Warbler	3	2	3	12	6	6
	Myioborus pictus	Painted Redstart	3	2	4	6	2	5
	Myioborus miniatus	Slate-throated Redstart	3	0	0	8	0	0
Cardinalidae	*Piranga flava*	Hepatic Tanager	0	1	1	0	1	2
	Piranga rubra	Summer Tanager	6	0	0	9	0	0
	Piranga ludoviciana	Western Tanager	2	1	0	2	1	0
	Piranga bidentata	Flame-colored Tanager	1	1	0	1	2	0
	Pheucticus melanocephalus	Black-headed Grosbeak	5	9	3	10	25	5
Thraupidae	*Diglossa baritula*	Cinnamon-bellied Flowerpiercer	1	2	0	5	3	0

Capítulo 3. Appendix 2. Surveyed green spaces characteristics. Densely-vegetated fractions are not shown. ID corresponds to the number displayed in figures 1 and 3.

						Habitat diversity		
ID	Name	Area (ha)	Shape index	Tree richness	Isolation (m)	Winter	Dry breeding	Wet breeding
1	Águilas	10.5	1.20	7	4351	0.419	0.467	0.214
2	Alabama	1.4	1.12	4	2904	0.609	0.459	0.532
3	Alameda	8.7	1.21	10	3622	0.569	0.549	0.644
4	Axiomatla	5.9	1.26	4	576	0.504	0.488	0.448
5	Barragán	1.4	1.18	3	2234	0.494	0.602	0.497
6	Bombilla	4.7	1.11	9	5585	0.684	0.523	0.666
7	Bosque de Tlalpan	105.6	1.10	16	0	0.514	0.473	0.144
8	Canarios	0.5	1.30	5	1675	0.452	0.503	0.571
9	Chapultepec	67.7	1.09	14	0	0.637	0.41	0.423
10	Ciudadela	1.2	1.22	8	2880	0.644	0.529	0.575
11	Desierto de los Leones	45.3	1.19	10	0	0.287	0.072	0.109
12	Dínamos	65.0	1.48	11	0	0.590	0.344	0.172
13	Deportivo Ajusco	2.3	1.27	4	108	0.662	0.675	0.727
14	Feria	47.6	1.06	23	0	0.648	0.673	0.674
15	Fuentes Brotantes	22.8	2.16	14	777	0.604	0.585	0.612
16	Gandhi	8.1	1.84	9	0	0.429	0.479	0.511
17	Hundido	8.6	1.21	19	3925	0.425	0.576	0.560
18	Islas	13.1	1.42	9	4018	0.661	0.673	0.689
19	Japón	11.7	1.64	9	3826	0.590	0.646	0.550
20	Jardín Morelos	0.8	1.12	8	1492	0.538	0.585	0.415
21	Lincoln	4.9	1.85	15	281	0.577	0.596	0.649
22	Lira	6.8	1.45	11	714	0.528	0.634	0.621
23	Lúdico Macondo	2.3	1.24	5	1797	0.546	0.574	0.549
24	México	7.5	1.15	13	1238	0.624	0.573	0.571
25	Pilares	3.5	1.17	9	4672	0.663	0.654	0.528
26	Pombo	0.5	1.13	6	2335	0.549	0.489	0.284
27	Ramón López Velarde	8.6	1.34	18	2459	0.585	0.617	0.558
28	Rosendo	1.4	1.19	3	2935	0.632	0.662	0.585
29	Santa María la Ribera	2.7	1.13	4	3482	0.557	0.562	0.547
30	Totoloapan	61.3	1.16	11	0	0.588	0.348	0.337
31	Venados	8.2	1.19	9	5573	0.699	0.399	0.539

32	Viveros	40.9	1.15	16	6556	0.552	0.427	0.454

Chapter 3. Appendix 3. Candidate global models to describe the variation of diversity indices and their standardized effect sizes for bird communities in green spaces during winter, early-breeding, and late-breeding seasons. Evidence ratio (ER) measures how much more likely the top-ranked model is the best model than model *i*. DVF: Densely vegetated fraction. Densely-vegetated fraction. For more details about explanatory variables check Table 1.

	Candidate models - Winter	**k**	**AIC$_c$**	**Δ_i**	**w$_i$**	**ER**	**R^2**
	Species richness†						
1	Isolation + Habitat diversity + Tree richness + Shape + DVF$_0$ + DVF$_{50}$	7	173.5	0.0	0.74		67.8
2	Isolation + Habitat diversity + Tree richness + Shape + DVF$_0$ + DVF$_{100}$	7	175.7	2.2	0.25	3.0	65.7
3	Isolation + Habitat diversity + Tree richness + Shape + DVF$_0$ + DVF$_{200}$	7	182.8	9.2	0.01	100.4	59.2
4	Isolation + Habitat diversity + Tree richness + Shape + DVF$_0$	6	203.9	30.3	0.00	3.8x10^6	36.3
	Faith's phylogenetic diversity						
1	Isolation + Habitat diversity + Tree richness + Shape + DVF$_0$ + DVF$_{50}$	8	410.8	0.0	0.48		56.4
2	Isolation + Habitat diversity + Tree richness + Shape + DVF$_0$ + DVF$_{100}$	8	411.0	0.2	0.44	1.1	56.1
3	Isolation + Habitat diversity + Tree richness + Shape + DVF$_0$ + DVF$_{200}$	8	414.5	3.6	0.08	6.1	51.0
4	Isolation + Habitat diversity + Tree richness + Shape + DVF$_0$	7	422.5	11.7	0.00	340.9	28.4
	SES Faith's phylogenetic diversity						
1	Isolation + Habitat diversity + Tree richness + Shape + DVF$_0$ + DVF$_{50}$	8	80.5	0.0	0.33		25.1
2	Isolation + Habitat diversity + Tree richness + Shape + DVF$_0$	7	80.6	0.2	0.30	1.1	15.2
3	Isolation + Habitat diversity + Tree richness + Shape + DVF$_0$ + DVF$_{100}$	8	81.3	0.8	0.22	1.5	23.1
4	Isolation + Habitat diversity + Tree richness + Shape + DVF$_0$ + DVF$_{200}$	8	81.9	1.5	0.16	2.1	21.4
	Functional richness						
1	Isolation + Habitat diversity + Tree richness + Shape + DVF$_0$ + DVF$_{100}$	8	-96.2	0.0	0.43		39.0
2	Isolation + Habitat diversity + Tree richness + Shape + DVF$_0$ + DVF$_{50}$	8	-95.8	0.5	0.34	1.3	38.1

3	Isolation + Habitat diversity + Tree richness + Shape + DVF_0 + DVF_{200}	8	-94.3	1.9	0.16	2.6	35.1
4	Isolation + Habitat diversity + Tree richness + Shape + DVF_0	7	-92.6	3.7	0.07	6.2	22.7
	SES Functional richness						
1	Isolation + Habitat diversity + Tree richness + Shape + DVF_0	7	95.1	0.0	0.64		22.2
2	Isolation + Habitat diversity + Tree richness + Shape + DVF_0 + DVF_{50}	8	98.4	3.3	0.12	5.2	23.1
3	Isolation + Habitat diversity + Tree richness + Shape + DVF_0 + DVF_{200}	8	98.4	3.3	0.12	5.3	23.0
4	Isolation + Habitat diversity + Tree richness + Shape + DVF_0 + DVF_{100}	8	98.5	3.4	0.12	5.6	22.8

† Over-dispersion was detected, so a quasi-GLM model was fitted.

	Candidate models – Dry breeding season	**k**	**AIC_c**	**Δ_i**	**w_i**	**ER**	**R^2**
	Species richness						
1	Isolation + Habitat diversity + Tree richness + Shape + DVF_0 + DVF_{100}	7	164.8	0.0	0.43		60.0
2	Isolation + Habitat diversity + Tree richness + Shape + DVF_0 + DVF_{50}	7	165.9	1.0	0.26	1.7	58.2
3	Isolation + Habitat diversity + Tree richness + Shape + DVF_0 + DVF_{200}	7	166.0	1.1	0.25	1.8	58.0
4	Isolation + Habitat diversity + Tree richness + Shape + DVF_0	6	168.7	3.8	0.06	6.7	47.3
	Faith's phylogenetic diversity						
1	Isolation + Habitat diversity + Tree richness + Shape + DVF_0 + DVF_{100}	8	384.0	0.0	0.32		51.3
2	Isolation + Habitat diversity + Tree richness + Shape + DVF_0 + DVF_{200}	8	384.1	0.2	0.29	1.1	51.0
3	Isolation + Habitat diversity + Tree richness + Shape + DVF_0 + DVF_{50}	8	384.7	0.8	0.22	1.5	50.1
4	Isolation + Habitat diversity + Tree richness + Shape + DVF_0	7	385.2	1.3	0.17	1.9	42.5
	SES Faith's phylogenetic diversity						
1	Isolation + Habitat diversity + Tree richness + Shape + DVF_0	7	72.1	0.0	0.56		15.1
2	Isolation + Habitat diversity + Tree richness + Shape + DVF_0 + DVF_{100}	8	74.4	2.4	0.17	3.3	19.0
3	Isolation + Habitat diversity + Tree richness + Shape + DVF_0 + DVF_{50}	8	74.9	2.8	0.14	4.1	17.7
4	Isolation + Habitat diversity + Tree richness + Shape + DVF_0 + DVF_{200}	8	75.1	3.0	0.13	4.4	17.3

	Functional richness						
1	Isolation + Habitat diversity + Tree richness + Shape + DVF_0 + DVF_{200}	8	-214.6	0.0	0.76		66.4
2	Isolation + Habitat diversity + Tree richness + Shape + DVF_0 + DVF_{100}	8	-211.7	3.0	0.17	4.4	62.8
3	Isolation + Habitat diversity + Tree richness + Shape + DVF_0 + DVF_{50}	8	-209.4	5.2	0.06	13.5	59.8
4	Isolation + Habitat diversity + Tree richness + Shape + DVF_0	7	-206.2	8.4	0.01	67.3	48.7
	SES Functional richness						
1	Isolation + Habitat diversity + Tree richness + Shape + DVF_0	7	80.0	0.0	0.49		29.7
2	Isolation + Habitat diversity + Tree richness + Shape + DVF_0 + DVF_{200}	8	81.2	1.2	0.27	1.8	35.9
3	Isolation + Habitat diversity + Tree richness + Shape + DVF_0 + DVF_{100}	8	82.6	2.6	0.13	3.7	32.7
4	Isolation + Habitat diversity + Tree richness + Shape + DVF_0 + DVF_{50}	8	82.9	2.9	0.11	4.3	32.0

	Candidate models – Wet breeding season	**k**	**AIC_c**	**Δ_i**	**w_i**	**ER**	**R^2**
	Species richness						
1	Isolation + Habitat diversity + Tree richness + Shape + DVF_0 + DVF_{50}	7	141.2	0.0	0.34		89.7
2	Isolation + Habitat diversity + Tree richness + Shape + DVF_0 + DVF_{100}	7	141.2	0.0	0.34	1.0	89.7
3	Isolation + Habitat diversity + Tree richness + Shape + DVF_0 + DVF_{200}	7	142.4	1.1	0.19	1.8	88.2
4	Isolation + Habitat diversity + Tree richness + Shape + DVF_0	6	143.1	1.9	0.13	2.5	82.5
	Faith's phylogenetic diversity						
1	Isolation + Habitat diversity + Tree richness + Shape + DVF_0 + DVF_{50}	8	354.8	0.0	0.61		79.9
2	Isolation + Habitat diversity + Tree richness + Shape + DVF_0 + DVF_{100}	8	356.3	1.5	0.29	2.1	82.2
3	Isolation + Habitat diversity + Tree richness + Shape + DVF_0 + DVF_{200}	8	358.9	4.1	0.08	7.8	81.3
4	Isolation + Habitat diversity + Tree richness + Shape + DVF_0	7	361.3	6.5	0.02	25.9	79.6
	SES Faith's phylogenetic diversity						
1	Isolation + Habitat diversity + Tree richness + Shape + DVF_0 + DVF_{200}	8	57.5	0.0	0.62		49.8
2	Isolation + Habitat diversity + Tree richness + Shape + DVF_0 + DVF_{100}	8	59.1	1.6	0.28	2.2	47.2

3	Isolation + Habitat diversity + Tree richness + Shape + DVF_0 + DVF_{50}	8	61.9	4.4	0.07	8.9	42.0
4	Isolation + Habitat diversity + Tree richness + Shape + DVF_0	7	63.7	6.1	0.03	21.5	30.2
	Functional richness						
1	Isolation + Habitat diversity + Tree richness + Shape + DVF_0 + DVF_{50}	8	-129.5	0.0	0.46		73.7
2	Isolation + Habitat diversity + Tree richness + Shape + DVF_0	7	-128.4	1.0	0.27	1.7	69.2
3	Isolation + Habitat diversity + Tree richness + Shape + DVF_0 + DVF_{100}	8	-127.7	1.7	0.19	2.4	72.2
4	Isolation + Habitat diversity + Tree richness + Shape + DVF_0 + DVF_{200}	8	-125.8	3.7	0.07	6.3	70.3
	SES Functional richness						
1	Isolation + Habitat diversity + Tree richness + Shape + DVF_0	7	95.0	0.0	0.42		14.4
2	Isolation + Habitat diversity + Tree richness + Shape + DVF_0 + DVF_{200}	8	95.9	0.9	0.27	1.5	22.3
3	Isolation + Habitat diversity + Tree richness + Shape + DVF_0 + DVF_{100}	8	96.6	1.6	0.19	2.2	20.4
4	Isolation + Habitat diversity + Tree richness + Shape + DVF_0 + DVF_{50}	8	97.5	2.5	0.12	3.5	17.9

Capítulo 3. Appendix 4. Zero-averaged estimates for predictors of taxonomic (TD), phylogenetic (PD), and functional diversity (FD) and the standardized effect sizes of phylogenetic diversity (SES PD) and functional richness (SES FD) at the site level. Models were fitted for winter (W), the dry breeding season (DB) and the wet breeding season (WB). Significant (bold) and marginally significant coefficients are shown. DVF: Densely vegetated fraction. Densely-vegetated fraction. For more details about explanatory variables check Table 1.

	DVF_0	Habitat Diversity	Isolation	Tree Richness	Shape	DVF_{50}	DVF_{100}	DVF_{200}
TD								
W				**0.183****		**0.364*****		
DB				0.132			**0.25****	
WB				**0.228****		**0.311***		
PD								
W						**129.24*****		
DB				48.514			68.454	
WB				**76.88*****		**65.00****		
SES PD								
W								
DB								
WB								
FD								
W								
DB				**0.002***				**0.004*****
WB				**0.021*****		**0.015****		
SES FD								
W								
DB	0.326							
WB								

Capítulo 3. Appendix 5. Zero-averaged estimates for predictors of taxonomic, phylogenetic, and functional diversity and the standardized effect sizes of phylogenetic diversity and functional richness at the sampling-point level. Models were fitted for all studied seasons. Significant (bold) and marginally significant coefficients are shown.

Season	Diameter at breast	Tree Height	Tree richness	Bare ground	Grass	Shrub	Tree	Equipment	Dog	People	Noise
Taxonomic diversity											
Winter				NA							NA
Dry breeding				NA							
Wet breeding†											
Phylogenetic diversity											
Winter				NA		**33.224****					NA
Dry breeding				NA							
Wet breeding			**36.666****								23.007
Standardized effect size of Phylogenetic diversity											
Winter				NA				**0.313****			NA
Dry breeding				NA				**0.316****			
Wet breeding											**0.294****
Functional diversity											
Winter				NA		**0.010****					NA
Dry breeding				NA						**-0.012*****	
Wet breeding							-0.009				
Standardized effect size of Functional diversity											
Winter				NA							NA
Dry breeding				NA				**-0.383*****			
Wet breeding							**-0.293****				