



# Species richness and ecological connectivity of the mammal communities in urban and peri-urban areas at Mexico City

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## Abstract

Urban expansion has emerged as a pervasive driver of biodiversity loss in Mexican cities due to shifts in landscape composition and configuration. Hence, the preservation of forest cover, green spaces and isolated trees assume a critical role in conserving the biodiversity within urban areas. We compared mammal assemblages across 520 sites in Mexico City examining the impact of local vegetation and site characteristics (e.g., patch size, isolation distance) on diversity patterns. Then, we used a generalized linear model to evaluate the relationship between mammal assemblages (including both alpha and beta diversities) and the level of structural and functional connectivity across the landscape. We collected 5,063 records of 38 species from the sampled sites. Mammalian richness ranged from 1 to 9 spp./site, and the mean of taxonomic and functional (Functional Distance [MFD]) alpha diversities were  $1.9 \pm 1.3$  spp./site and  $0.41 \pm 0.60$  MFD/site, respectively. The average size of habitat fragments was 0.41 hectares, and the percent forest cover per site ranged from 0.5 to 100%. Both species and functional alpha diversities were different among the land-use and vegetation types (including protected vs. non-protected areas). There was a clear distinction ( $\beta$  value  $> 0.40$ ) in species composition across the landscape, which was positively related to geographical distance and negatively related to connectivity between sites. We identified four main patches especially important to connectivity across the city and argue that conserving them is critical for restoring key components of biodiversity and ecosystem services in urban and peri-urban areas of Mexico City.

**Keywords** Mammal assemblages · Functional connectivity · Land-use scenarios · Structural connectivity · Urban ecology

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## Introduction

In recent decades, we have witnessed an unprecedented acceleration of global population growth and urban expansion. The increase and concentration of the world's population in cities is such that by 2050, it is expected that 70% of the world's people will inhabit urban centers (Smith et al. 2018; Forte et al. 2019). Rapid urbanization has been identified as one of the most important drivers of biodiversity loss, mainly due to the transformation and accelerated change of natural and peri-urban environments, which causes drastic changes in the composition and configuration of landscapes (Grimm et al. 2008; Maxwell et al. 2016). Urbanization—which is often unplanned—poses a considerable threat to the richness and abundance of local species, leading to the possible extirpation or extinction of various wildlife taxa (Dirzo et al. 2014; Newbold et al. 2015). Therefore, there is an urgent need to incorporate biodiversity conservation programs into decision-making processes for urban planning to buffer the negative impacts on biota. Such efforts could contribute to not only providing suitable natural habitat for these species within and on the outskirts of urban areas for species but also ensure the resilience of biota in these environments (Frantzeskaki et al. 2017; Zhang and Li 2018).

Although many urban environments are adverse for wildlife, recent evidence demonstrates that even within the urban matrix there are important green spaces (such as gardens, parks, sidewalks, lawns, empty lots, etc.) that play a crucial role as refuges for biodiversity, feeding niches, and maintaining ecological connectivity (e.g., MacGregor-Fors 2016; Lepczyk et al. 2017). However, these sites offer low-quality habitat for most species, and are generally too small to sustain viable populations. Thus, the survival of wild species depends on their ability to move among patches within urban areas in search of shelter, food, and other resources (Bergerot et al. 2013; Ofori et al. 2018). In this context, understanding the composition, quality, and connectivity of these vegetation patches becomes paramount for biodiversity conservation (Ofori et al. 2018). This information serves as a critical prerequisite to improve decisions in urban planning and promote urban development that facilitates long-term conservation of biodiversity (Löfvenhaft et al. 2004). This is critical in urban and peri-urban areas where the size of green spaces has considerably diminished (Yu et al. 2012; LaPoint et al. 2015; Zambrano et al. 2022).

Mammals are an excellent wildlife group for studying responses of biota to urbanization for several reasons. Mammals are highly sensitive to human-modified landscapes, as their occurrence and assemblages depend on the composition and structure of natural vegetation (Garden et al. 2007; McKinney 2008; Chávez and Ceballos 2009). The current status of mammalian diversity is troubling, with over

a quarter of terrestrial mammal species facing threats of extinction and nearly half of all species showing declining trends (Schipper et al. 2008; IUCN 2021). Moreover, mammals play critical ecological roles such as seed dispersion, herbivory, and pest control, even in urban areas (Ramos-Lara and Gómez-Ortiz 2019). Also, the life-history and ecological traits (including morphological data, range size, trophic levels, vulnerability risk, etc.) of several species worldwide are well-documented (Fischer et al. 2012; Weiss and Ray 2019; Hantak et al. 2021), offering a global template for testing hypotheses and exploring the structure and functioning of this biodiversity component. Lastly, mammals simultaneously fulfill the roles of flagship, umbrella, indicator, and keystone species (Lindenmayer and Westgate 2020; Zhang et al. 2020). Therefore, monitoring populations of mammals and incorporating them into conservation policies is of paramount importance.

Although urban ecology emerged as a discipline in the 1980s, studies focusing on these topics in Latin American cities are scarce. When such studies do exist, they are often limited to taxonomic listings in specific green spaces (e.g., parks and squares) and focus mainly on birds (McKinney 2008; MacGregor-Fors and Escobar-Ibáñez 2017). This appears to be the case for Mexico City (abbreviated CDMX, for its Spanish name), which boasts high levels of species richness across several biological groups (Rivera and Espinosa 2007; CONABIO and SEDEMA 2016; Pacheco-Muñoz et al. 2022). While there has been an increase in taxonomic and ecological knowledge of mammal communities over the past few decades (e.g., Ceballos and Galindo 1984; Ramírez-Pulido et al. 1986, 2005; Chávez and Ceballos 1998; Villa and Cervantes 2003; Navarro-Frías et al. 2007; Hortelano-Moncada and Cervantes 2011, 2016; Hortelano-Moncada et al. 2021), few studies have evaluated the distribution patterns of wild mammals and assessed potential richness in CDMX as a whole (but see Navarro-Frías et al. 2007). More information and integrative studies considering both local and regional data are needed to better comprehend the magnitude of the effect of urbanization and landscape transformation on this aspect of Mexican biodiversity (CONABIO and SEDEMA 2021).

In this study, we aim to characterize the spatial patterns of alpha and beta diversity of mammals across the urban and peri-urban areas of CDMX, including taxonomic and functional dimensions of diversity. Specifically, we seek to answer the following questions: (1) Do mammal assemblages differ across the sites that comprise the urban and peri-urban landscape? and (2) What are the levels of structural and functional connectivity for mammal assemblages across the landscape? Urbanization acts as an ecological filter for biodiversity and consequently we hypothesize that physically closer sites will have more similar mammalian

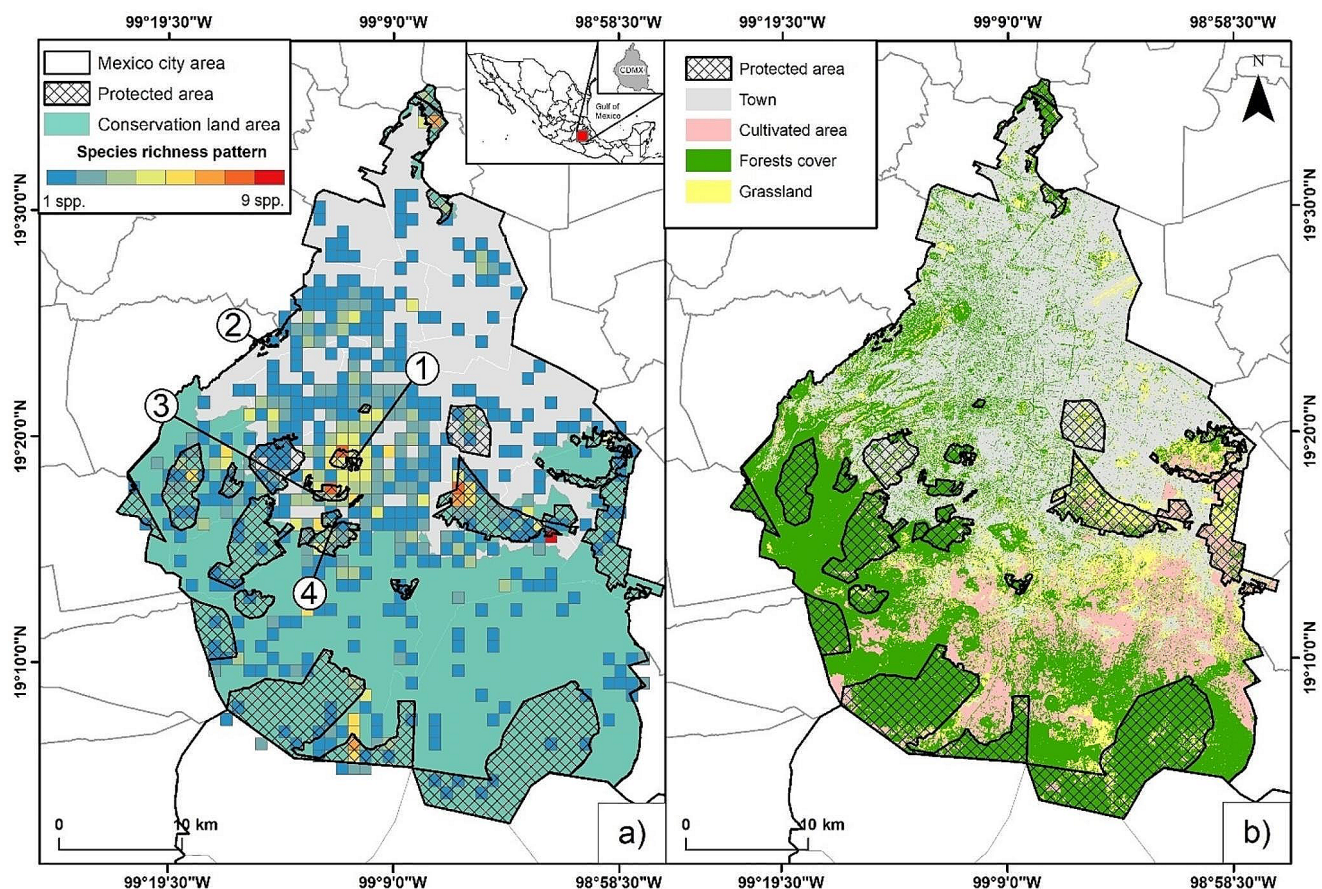
assemblages and display similar functional composition. A better understanding of these patterns will aid in determining management priorities for threatened species and inform land development planning and landscape design to conserve urban biodiversity more effectively amidst ongoing and widespread environmental change.

## Methods

### Study area

The CDMX (19°26' N – 99°8' W; Fig. 1a) is one of the largest cities in the world, spanning an area of 1,485 km<sup>2</sup> (0.1% of the area of the country). This city is densely populated, with around 9,210,000 inhabitants (INEGI, 2021). It is located in the Trans-Mexican Volcanic Belt biogeographic province in central Mexico and has an elevation of between

2,250 and 3,950 masl (CONABIO and SEDEMA 2021). CDMX has a tropical to temperate climate with a relatively small annual temperature range (5–25°C) and precipitation (600–1500 mm/year) showing three marked seasons along the year: cold dry (Nov–Feb), warm dry (Mar–Apr), and the rainy season (May–Oct; INEGI 2017). This geographical area of CDMX has been recognized as an important conservation area due to its high levels of diversity and endemism; it contains 1,600 documented species of plants, 3,851 species of invertebrates and 350 vertebrate species (CONABIO and SEDEMA 2016). Therefore, more than 50% of its territory is catalogued as “Conservation Land” by the national environmental authorities (Fig. 1). Moreover, CDMX contains a diverse mosaic of landscapes dominated by urban areas and greenspaces (48%), but also including oak and pine forests, xerophytic scrub, cloud forest, natural grassland, and wetlands (36%), as well as peri-urban and agricultural areas (16%). CDMX contains 26 natural protected areas,



**Fig. 1** Geographical position of Mexico City (CDMX) showing (a) the taxonomic diversity pattern observed for mammals ( $n = 38$  spp.) across the 520 sampled sites, indicating the four critical green spaces (1) “Pedregal San Ángel” Ecological Reserve (REPSA); (2) the Chapulte-

pec urban park; (3) the “Bosque de Tlalpan”; and (4) the CDMX Ecological Park; and (b) the spatial arrangement of land use and vegetation cover in the landscape

encompassing approximately 26,700 hectares (CONABIO and SEDEMA 2016). Moreover, it is important to highlight that many urban parks and gardens across CDMX maintain biophysical and scenic characteristics of the natural vegetation of the area, despite the high degree of anthropogenic change (CONABIO and SEDEMA 2021).

### Species list and historical data

To obtain the species list of mammals and their historical occurrence records across CDMX, we compiled information from four sources: (a) online collaborative public databases (i.e., Global Biodiversity Information Facility's [GBIF 2022], Mammal Networked Information System [MANIS; Wieczorek 2001], and iNaturalista [2022]); (b) the Informatics Unit for Biodiversity (UNIBIO, 2006); (c) the specialized literature (Ceballos and Galindo 1984; Ramírez-Pulido et al. 1986; Chávez and Ceballos 1998; Villa and Cervantes 2003; Navarro-Frías et al. 2007; Hortelano-Moncada and Cervantes 2011, 2016; Hortelano-Moncada et al. 2021); and (d) fieldwork and monitoring projects performed by our research group between March 2019 and July 2022) using 18 camera trapping stations across the urban and peri-urban areas in southern CDMX. Moreover, because citizen science makes an important contribution to ecological databases (e.g., Poisson et al. 2020; Sun et al. 2021), we complemented our data with information obtained from the online form (<https://forms.gle/5YYCsJheqM5eQ3sBA>; May 7th to December 27, 2022) implemented to CDMX's broad public. Further integrating citizen science data with other datasets and datatypes can improve population estimates and expand the spatiotemporal extent of inference (Sun et al. 2021). For data collected using the online survey form, we only considered distribution records that included a precise location and photograph of the specimen.

Occurrence data was then filtered and cleaned. We used the R library *CoordinateCleaner* (Zizka et al. 2019) to remove unsuitable or unreliable records, eliminating data that lacked collection location and/or year. We also eliminated records that were duplicated across sources, retaining only information corresponding to unique localities (i.e., sites with a minimum separation of  $\sim 500 \text{ m}^2$ ) to avoid spatial overrepresentation (Roubicek et al. 2010; Boria et al. 2014). Each locality was verified and georeferenced when necessary to a precision of at least 100 m using Google Earth Pro and LONGITUDE (<http://www.longitudemaps.com/>). For each case, geographic coordinates were transformed to decimal degrees based on the WGS84 datum. Localities whose coordinates had less than four decimal places were disregarded. We only included information from 2016 to 2022 to provide a contemporary representation of mammal assemblages, i.e., those that could still be present today

(CONABIO and SEDEMA 2016). All families and species names (see Table 1) followed the nomenclature of Ramírez-Pulido et al. (2005) and Hortelano-Moncada & Cervantes (2016).

### Taxonomic and functional diversity patterns of mammals' assemblages

We constructed a species presence-absence matrix (PAM) that overlapped the occurrence records of each taxon on an equal-area grid with a  $1 \times 1 \text{ km}$  spatial resolution (Fig. 1a). To assess how well the mammal assemblages are captured in our dataset we calculated expected species accumulation curves using a sample-based rarefaction method (Colwell et al. 2012) based on three adjustment models (Chao, Jackknife 1, and Bootstrap) as implemented in the BiodiversityR library (Oksanen et al. 2018). Then, we calculated spatial taxonomic alpha diversity by adding to each focal cell the number of its recorded species. Spatial taxonomic beta diversity was calculated using the Sørensen-based multiple-site index to determine the total of species, nested and turnover (Baselga and Orme 2012).

We decided to include a functional diversity metric because this quantifies different organismal traits (e.g., morphological, physiological, behavioral, or phenotypical traits) that intervene directly or indirectly in ecosystem processes and influence responses to both biotic and abiotic environmental conditions (Gómez-Ortiz and Moreno 2017; Weiss and Ray 2019). To do this, we determined the spatial functional alpha diversity based on the mean functional distance (MFD) among all pairs of species using a functional dendrogram, which represented the ecological traits distance among taxa (Hidasi-Neto et al. 2019). The final dendrogram of MFD was built using Gower's distance and Unweighted Pair Group Method with Arithmetic Mean (UPGMA). For each species we included information from ecological traits related to resource and habitat use (Ceballos and Oliva 2005; Aranda 2012; Fischer et al. 2012; González-Maya et al. 2016; SEMARNAT 2019; Weiss and Ray 2019; Hantak et al. 2021; IUCN 2021): body mass, size, foraging strata/diet, habit, activity period, conservation status, CDMX-restricted level (i.e., proportion of distribution into CDMX), geographic range, and abundance values. Details on general methods used to obtain data for ecological traits are included in Table S1. For the spatial functional beta diversity, we used the same process as for the taxonomic beta, except that we implemented the turnover partition of the PhyloSor index (Baselga and Orme 2012). All analyses were performed with the "*betaGrid*" function in R (available at: <http://rfunctions.blogspot.com/>). Finally, we generated maps for both spatial alpha and beta diversities.

## Land use and land cover map (LULC)

A land-use and land cover map (Fig. 1b) was generated from a Sentinel-2 satellite image acquired in March 2022 obtained from the US Geological Survey – USGS Earth Explorer (<https://earthexplorer.usgs.gov/>). The image classification was performed using an object-based approach and the Random Trees classifier from Trimble eCognition® Developer 9.0. Four categories of land use and vegetation were identified: town or urban, grassland, cultivated areas and forest cover. However, these were ultimately reclassified into two broad categories (forest cover vs. deforested) in order to specifically evaluate the effect of forest cover on biodiversity. To guarantee precise classification, the training procedure adhered to the guidelines of Campbell et al. (2015), which entail the selection of 100 sample units per land use category and refining the outcomes through visual scrutiny (i.e., testing process). In this study, we took a total of 400 samples, using 200 for the classification and 200 for validation. These samples were obtained through field visits and recent Google Earth images, locating plots with a minimum of 60×60 m of homogeneous vegetation. The final maps were validated using the 100 samples mentioned earlier, and an area-based error matrix and Kappa index were generated for each classification, following the methodology proposed by Congalton and Green (2009).

## Landscape analysis (structure and functional)

To describe the diversity and spatial arrangement of landscape elements in CDMX, we utilized a 1×1 km grid (Fig. 1) to analyze the landscape structure and functionality. Fragstats v4.2.1 (McGarigal et al. 2012) was employed to calculate class area (CA) and percentage of landscape occupied by each class (PLAND). These metrics serve as fundamental measures of landscape composition, indicating the proportion of the landscape occupied by each class type. Using these results, we reclassified the LULC map (see above) into four categories based on percent forest cover: deforested (<10% forest cover), low cover (10–25% forest cover), intermediate cover (25–65% forest cover), and high cover (>65% forest cover). Furthermore, we conducted Kruskal-Wallis tests to determine whether there were differences in species richness among the four land use categories identified herein but also between protected and non-protected areas in CDMX. The shapefile of the boundaries of terrestrial protected areas were obtained from CONANP (available on: [http://sig.conanp.gob.mx/website/pagsig/info\\_shape.htm](http://sig.conanp.gob.mx/website/pagsig/info_shape.htm)) and SEDEMA (<https://datos.cdmx.gob.mx/dataset/areas-naturales-protegidas>) websites.

Functional connectivity evaluates how species respond to the physical structure of the landscape (Alonso et al.

2017). The degree to which the landscape facilitates the species' movement between elements defines the degree of connectivity and is directly associated with how the location, shape, and size of forest patches affect the abundance and distribution of species in a landscape. We applied the connectivity probability (PC) index to integrate the area of habitat patches and their connectivity using graph theory to measure landscape connectivity. In general terms, the PC increases with improved connectivity and has a bounded range of variation from 0 to 1. The PC index was calculated using Conefor Software 2.6 (Saura and Torné 2009) for six threshold distances (50, 100, 250, 500, 750 and 1,000 m), each corresponding to the dispersal potential of different taxonomic groups. In our study, distances greater than 1,000 m were not utilized because if an organism can move 500 m or more, the species already has access to all of the patches of natural habitat within the city. Overall, two scenarios were generated: (a) connect the central part of the city with the most important remnants of the periphery and (b) connect the four internal CDMX priority sites (obtained through the PC) for the adequacy of internal corridors. We used a generalized linear model to assess the effect of connectivity (i.e., the PC values) and geographical distance (based on Euclidean measures) among forest patches on the species richness (alpha diversity) and taxonomic/functional composition (beta diversity) throughout the landscape (Muench 2017).

To evaluate the potential corridors, we used the Gnarly and Linkage Mapper tools (McRae and Kavanagh 2011; McRae 2012). Gnarly Landscape Utilities is an ArcGIS toolbox that generates resistance maps (using an index from 0 to 100) to assess habitat quality and territorial friction given species' dispersal capacity. Here, we used five variables to assess the habitat quality and territorial friction in response to species displacement (i.e., resistance map): land use and land cover map, distance to protected natural areas, elevation, distance to water bodies, and slope (Von Thaden et al. 2021). We built the raster of distance from protected natural areas and water bodies using the “Euclidean distance” function in ArcMap 10.8. (ESRI 2010), based on the information provided by the INEGI (<https://www.inegi.org.mx/>). Finally, we determined the routes with the lowest cost of movement (cost weighted) between the core areas, which corresponded to the areas of high importance for functional connectivity (i.e., grids with PC values equal or greater than 0.4).

## Results

### Species list and historical data

Based on occurrence data from 2016 to 2022, we obtained a total of 5,063 validated records for 520 sites that represent 30.3% of the city's surface area (Fig. 1a). Our records contained 38 species of mammals (45% of the total species historically recorded in CDMX), grouped into eight orders and 17 families (Table 1). The rarefaction curves of the three adjustment models reached asymptotes (Fig. S1), indicating that few new species remain to be recorded in all areas. The estimators of total species number predicted a richness of 42 (Bootstrap) to 46 (Chao and Jackknife) species, indicating that the attained sampling effort was able to record 82.6–90.5% of the whole biota. Thus, the inventories are sufficiently complete for these areas.

The group with the highest number of species reported was that of rodents ( $n=11$ ), followed by carnivores ( $n=9$ ) and chiropterans ( $n=8$ ). Of the total number of species, six are endemic to central Mexico: the volcano rabbit (*Romerolagus diazi*), the cuinique squirrel (*Notocitellus adocetus*), the cotton tail rabbit (*Sylvilagus cunicularius*), the tailed shrew (*Sorex ventralis*), the motocle squirrel (*Ictidomys mexicanus*) and the small-eared shrew (*Cryptotis alticola*). We also reported the occurrence of flagship species such as cacomixtles (*Bassariscus astutus*) and gray fox (*Urocyon cinereoargenteus*).

### Landscape properties: structural and functional connectivity

The overall accuracy and Kappa index (K) of our final land-use and land cover map (LULC) map were estimated to be 92% and 0.9, respectively. Our mapping revealed that 44% of the area of CDMX is covered by forest, mostly in the southern part of the city (Fig. 1b). Human settlements occupy 33% of the total area, crops cover 12%, and grassland accounts for 11%. The two largest forest fragments in CDMX measured 48,015 and 1,244 hectares, respectively; but the average size of fragments in CDMX is 0.41 hectares. Grid cells varied in forest cover from 0.5 to 100% in the study area (Fig. 2). Landscape-level metrics showed that 9.2% of the 520 locations with mammal records were in deforested areas, 24.8% in low cover forest, 42.1% in intermediate cover forest, and 23.9% in high forest cover (Table 1; Fig. 2).

The overall mean PC index value was 0.26. On average, the PC value between the threshold dispersal distance of 50 and 1,000 m increased by 65%. The PC value increase is inversely proportional (i.e. at greater distances, the increase in connectivity is less) to the threshold distance resulting in

a PC value increase of 35% between 50 and 100 m, but values between 750 and 1,000 only increase PC value by 2% (Fig. 2). We observed that for all threshold distances used in our models, the highest PC values are primarily concentrated in the southern, southwestern, and western parts of CDMX (Fig. 2b-c); which corresponds closely to the prevalence of existing vegetation.

### Spatial biodiversity patterns for mammals across CDMX landscape

The number of sites at which a species was reported ranged from 1 to 280, with 31.6% of the species ( $n=12$ ) reported at only one or two sites. About 10.5% ( $n=4$ ) of the species were recorded in at least 25% of the study area, while 13.2% ( $n=5$ ) of species were reported in 5–15% of sites. Most of the mammal species were recorded in less than 5% of the sites. According to our map, 57.8% of sites had a single species reported, 36.8% of the sites had 2–4 species, and only 5.4% of sites had five or more species.

Mammal species richness ranged from 1 to 9 spp./site, with an average of  $1.9 \pm 1.3$  species per sampling site. The functional richness per site was  $0.41 \pm 0.60$  MFD. We found a positive correlation between taxonomic species richness and functional richness of communities by site ( $r=0.66$ ,  $t=19.89$ ,  $P<0.001$ ). Based on the Kruskal-Wallis test, the number of species and functional richness differed among the four vegetation types (Taxonomic: *Chi-squared*: 26.506,  $df=3$ ,  $P<0.001$  and Functional: *Chi-squared*: 44.234,  $df=3$ ,  $P<0.001$ ) (see Table 2). There was also a difference in taxonomic and functional diversity between protected and non-protected areas across CDMX (Taxonomic: *Chi-squared*: 18.582,  $df=1$ ,  $P=0.021$  and Functional: *Chi-squared*: 31.969,  $df=1$ ,  $P<0.001$ ).

The overall mean beta diversity values were  $0.73 \pm 0.35$  for taxonomic diversity and  $0.48 \pm 0.62$  for functional diversity, indicating a clear distinction in composition among CDMX sites. Even within the same vegetation type, beta diversity values were high ( $>0.67$ ) (Table 2). Moreover, the  $\beta$  diversity among vegetation types ranged from 0.06 (intermediate cover vs. high cover) to 0.36 (deforested vs. high cover). There was a mean  $\beta$  value of 0.40 between protected and non-protected areas (Table 2). Moreover, generalized linear models showed that beta diversity was positively related to geographical distance ( $R^2=0.059$ ,  $F=0.00168$ ,  $P<0.001$ ) and negatively related to connectivity values ( $R^2=0.073$ ,  $F=46.13$ ,  $P<0.001$ ) between sites.

We detected four hotspots (i.e., sites with high species richness) where forests cover contributed significantly ( $PC>0.40$ ) to connectivity despite their apparent fragmentation (Fig. 2). Such areas were distributed mainly across the center of the city, including the “Pedregal San Ángel”

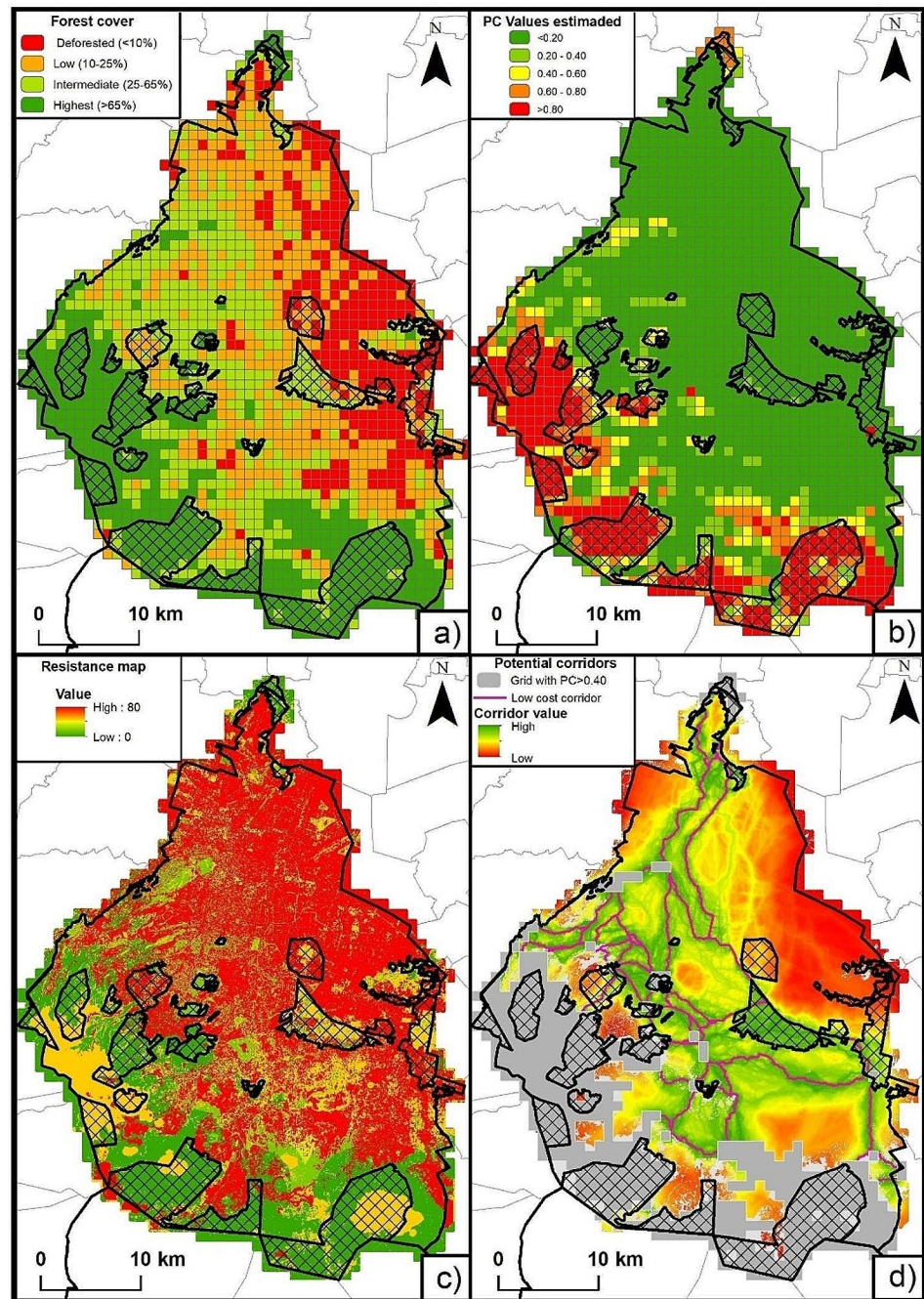
**Table 1** List of presence records of mammal species recorded from 2016 through 2022 across 520 sites in Mexico City. For each species, we show the taxonomic designation with notes on conservation status and distribution according to national (SEMARNAT, 2019) and international (IUCN, 2021) checklists

Order and Family	Species	IUCN category	National conservation status	Distribution
<b>ARTIODACTYLA</b>				
Cervidae	<i>Odocoileus virginianus</i>	Least Concern	Not included	North America
<b>CARNIVORA</b>				
Canidae	<i>Canis latrans</i>	Least Concern	Not included	North America
Felidae	<i>Lynx rufus</i>	Least Concern	Not included	North America
Mephitidae	<i>Conepatus leuconotus</i>	Least Concern	Not included	North America
Procyonidae	<i>Bassariscus atutus</i>	Least Concern	Not included	North America
Mephitidae	<i>Mephitis macroura</i>	Least Concern	Not included	Widely-Distributed
Mustelidae	<i>Mustela frenata</i>	Least Concern	Not included	Widely-Distributed
Procyonidae	<i>Procyon lotor</i>	Least Concern	Not included	North America
Mephitidae	<i>Spilogale angustifrons</i>	Least Concern	Not included	North America
Canidae	<i>Urocyon cinereoargenteus</i>	Least Concern	Not included	North America
<b>CHIROPTERA</b>				
Molossidae	<i>Tadarida brasiliensis</i>	Least Concern	Not included	Widely-Distributed
Phyllostomidae	<i>Anoura geoffroyi</i>	Least Concern	Not included	Widely-Distributed
Phyllostomidae	<i>Choeronycteris mexicana</i>	Near Threatened	Endangered	North America
Phyllostomidae	<i>Leptonycteris yerbabuenae</i>	Near Threatened	Not included	North America
Vespertilionidae	<i>Corynorhinus mexicanus</i>	Near Threatened	Not included	Endemic
Vespertilionidae	<i>Lasiurus blossevillii</i>	Least Concern	Not included	Widely-Distributed
Vespertilionidae	<i>Lasiurus cinereus</i>	Least Concern	Not included	Widely-Distributed
Vespertilionidae	<i>Myotis californicus</i>	Least Concern	Not included	Widely-Distributed
<b>CINGULATA</b>				
Dasypodidae	<i>Dasypus novemcinctus</i>	Least Concern	Not included	Widely-Distributed
<b>DIDELPHIMORPHIA</b>				
Didelphidae	<i>Didelphis virginiana</i>	Least Concern	Not included	Widely-Distributed
<b>EULIPOTYPHLA</b>				
Soricidae	<i>Cryptotis alticola</i>	Data Deficient	Special protection	Endemic
Soricidae	<i>Cryptotis parvus</i>	Least Concern	Special protection	Mesoamerica
Soricidae	<i>Sorex sausseri</i>	Least Concern	Special protection	Mesoamerica
Soricidae	<i>Sorex ventralis</i>	Least Concern	Not included	Endemic
<b>LAGOMORPHA</b>				
Leporidae	<i>Romerolagus diazi</i>	Endanger	Endanger	Endemic
Leporidae	<i>Sylvilagus cunicularius</i>	Least Concern	Not included	Endemic
Leporidae	<i>Sylvilagus floridanus</i>	Least Concern	Not included	Widely-Distributed
<b>RODENTIA</b>				
Cricetidae	<i>Baiomys taylori</i>	Least Concern	Not included	North America
Cricetidae	<i>Microtus mexicanus</i>	Least Concern	Not included	North America
Cricetidae	<i>Peromyscus gratus</i>	Least Concern	Not included	North America
Cricetidae	<i>Peromyscus maniculatus</i>	Least Concern	Not included	North America
Cricetidae	<i>Peromyscus melanotis</i>	Least Concern	Not included	North America
Geomyidae	<i>Cratogeomys merriami</i>	Least Concern	Not included	Endemic
Geomyidae	<i>Thomomys umbrinus</i>	Least Concern	Not included	North America
Sciuridae	<i>Ictidomys mexicanus</i>	Least Concern	Not included	Endemic
Sciuridae	<i>Notocitellus adocetus</i>	Least Concern	Not included	Endemic
Sciuridae	<i>Otospermophilus variegatus</i>	Least Concern	Not included	North America
Sciuridae	<i>Sciurus aureogaster</i>	Least Concern	Not included	Mesoamerica

Ecological Reserve (REPSA, for its Spanish name), the Chapultepec urban park, the “Bosque de Tlapan”, and the Ecological Park of CDMX (Fig. 1). In addition, our results showed that the highest resistance areas (values up to 80) in the CDMX landscape correspond mainly to urbanized sites

across the northeast. However, in the southern zone, where conservation areas are located, there are also areas with resistance values of 40–50 (Fig. 2c). These sites correspond particularly to those with steep slopes and higher elevations. Finally, we identified 20 key potential corridors crossing

**Fig. 2** (a) Estimated percentages of forest cover in each sampled grid; (b) the probability of connectivity (i.e., PC) values for each grid using 500 m as the threshold distance; (c) the resistance map and (d) potential corridors and the low-cost corridor (purple line) that allow connecting the grids with values higher to 0.40



CDMX from north to south (Fig. 2d) and connecting fragments at the lowest cost, with a total length of 278 km.

## Discussion

The distribution, community composition, and functional role of mammals has not been widely explored in CDMX (Hortelano-Moncada and Cervantes 2011; Hortelano-Moncada et al. 2021). This leaves a critical knowledge

gap because deforestation and land use change continue to occur at a rapid rate in this large and important city, putting the remaining mammal biodiversity (and the ecosystem services they provide) at risk (CONABIO and SEDEMA 2021). From this perspective, our study provides important insights into the consequences of urban environments on the distribution of mammal species, but also on species richness and community composition. Although our estimations were based on contemporary records of the species and landscape scale analyses—possible underestimating



**Table 2** Summary of alpha and beta taxonomic and functional biodiversity patterns recorded across the 520 sampled sites for mammal communities ( $n=38$  spp.) in the Mexico City landscape

	Number of sites	Taxonomic diversity		Functional diversity	
		A (spp./site)	$\beta$	A (MFD/site)	$\beta$
<b>LAND-USE AND VEGETATION TYPE*</b>					
Deforested (< 10% cover)	48	$1.1 \pm 0.7^a$	$0.78 \pm 0.15$	$0.09 \pm 0.34^a$	$0.14 \pm 0.41$
Low cover (10–25% cover)	129	$1.5 \pm 1.1^b$	$0.72 \pm 0.15$	$0.31 \pm 0.55^b$	$0.36 \pm 0.58$
Intermediate forest (25–65% cover)	219	$1.9 \pm 1.3^c$	$0.67 \pm 0.13$	$0.58 \pm 0.64^c$	$0.59 \pm 0.64$
Highest cover (> 65% cover)	124	$2.0 \pm 1.4^c$	$0.79 \pm 0.17$	$0.69 \pm 0.14^c$	$0.53 \pm 0.63$
<b>CONSERVATION AREA TYPE*</b>					
Protected areas	175	$2.1 \pm 1.6^a$	$0.81 \pm 0.16$	$0.72 \pm 0.15^a$	$0.46 \pm 0.60$
Non-Protected areas	345	$1.4 \pm 0.9^b$	$0.71 \pm 0.15$	$0.62 \pm 0.12^b$	$0.49 \pm 0.63$
<b>GENERAL PATTERN</b>	520	$1.9 \pm 1.3$	$0.73 \pm 0.35$	$0.41 \pm 0.60$	$0.48 \pm 0.62$

\*Letters correspond to identified groups based on the non-parametric Dunn test

the diversity patterns found— this approach allowed us to determine the degree of structural connectivity and quantify species' responses to the urban landscape (De Knecht et al. 2010; Bradfield et al. 2022). In fact, our findings underscore the significance of implementing ecological corridors to ensure the conservation of biota and ecological resilience in the city (Garden et al. 2007; Yu et al. 2012; Ofori et al. 2018). These measures are crucial for promoting more sustainable and wildlife-friendly cities while addressing the challenges posed by urbanization (LaPoint et al. 2015; Zhang and Li 2018).

As we expected, our results showed a direct association between vegetation cover/type and the taxonomic and functional diversity of wild mammals across the CDMX landscape: there are higher diversity values per site where there is intermediate and high vegetation coverage compared to deforested and low-coverage sites (Table 2). We therefore argue that peri-urban sites, while influenced by human activities, still retain some semblance of natural habitat, supporting a more diverse array of mammalian fauna (Magioli et al. 2016). In fact, we observed higher alpha diversity and greater species heterogeneity in the southern and southwestern region (e.g. in sites such as REPSA and Bosque de Tlalpan) of the city (Fig. 1), while the central and eastern areas of the city had few mammal records and greater degrees of biotic homogenization in both taxonomic and functional diversity. This main result can be explained by the availability of food, shelter, and connectivity provided by vegetation in urban areas (Ramos-Lara and Gómez-Ortiz 2019). Such trends are consistent with prior results that have documented shifts in mammal communities in response to urbanization across different cities in the world (Clavero and Brotons 2010; Soulsbury et al. 2010; Moll et al. 2020).

Our findings on functional alpha diversity also reveal how the ecological roles and traits of species vary across different environmental contexts in CDMX. Functional diversity per site was lower in deforested areas compared to areas

of intermediate and high vegetation cover (Fig. 2; Table 2). These results suggest that the conversion of natural habitats to urban developments and intensive agriculture in CDMX has repercussions not only in terms of the loss of species, but also for the loss of functional traits and reduction of functional redundancy, thus affecting ecosystem functioning (Clavel et al. 2011; Magioli et al. 2016). The stark contrast in diversity values between protected and non-protected areas highlights the efficacy of these conservation areas in protecting both species and functional traits. Our results show that the maintenance of functional diversity in urban environments depends on relationships between ecological traits of species (e.g., dispersal distance, home range size, body size, abundance, and diet) and the degree of fragmentation/alteration in the landscape (Miguet et al. 2016; Magioli et al. 2016). For example, there were areas within the urban zone with low vegetation cover (< 25%) that still had a significant number of rodent records, primarily due to the presence of two species of squirrels (*Sciurus aerogaster* and *Otospermophilus variegatus*) that can thrive in areas with limited vegetation (Ceballos and Arroyo-Cabrales 2012). The species *O. variegatus* is fossorial, so it can be found in areas with bare soil or little tree cover, while *S. aerogaster* was associated with green areas such as parks and gardens that serve as refuges for diversity in urban environments (Lepczyk et al. 2017).

From this perspective, in CDMX there is a clear distinction in the composition of species and ecological functionality of mammal assemblages that is determined by forest cover. It is important to note that more than half of the species found in highly urbanized, low-vegetation-cover areas were generalists that show behavioral adaptations to urban environments, such as the cacomixtle (*B. astutus*), opossum (*D. virginiana*) and squirrels (e.g., *O. variegatus*). These species can benefit from altered resource availability and novel ecological niches, obtaining food and nesting/refuge areas throughout sites with medium to high degrees of

disturbance (Dotta and Verdade 2011; Magioli et al. 2016). However, species that are more sensitive to fragmentation (e.g., *U. cinereoargenteus* and *C. merriami*) are lost as urbanization increases due to their size, diet, and requirements for conserved habitat (Jackson and Fahrig 2012; Miguet et al. 2016). Therefore, if fragmentation increases across the landscape in the near future, only generalist species with extensive movement capabilities that allow them to climb trees or buildings are likely to persist (McKinney 2008; Weiss and Ray 2019).

Based on these results, we argue that conservation and urban planning measures that focus on ecological connectivity across the landscape are required to promote ecological resilience in the face of ongoing global change (i.e. climate and land-uses) in urban environments such as CDMX (Fagan and Holmes 2006; Heller and Zavaleta 2009; Elsen et al. 2020). However, the movement capability and vulnerability patterns of species in urban environments must be considered in urban planning efforts (Merenlender et al. 2022). For example, we estimated that distance among main vegetation patches (i.e., structural connectivity) must not be higher than 100 m to avoid the loss of specialist mammals in the CDMX landscape. Thus, our results highlight the pivotal role of strategic green infrastructure planning in fostering landscape connectivity within CDMX. However, future research must explore various mechanisms and scales of species-landscape relationships, including patch foraging behavior and dispersal capabilities (Jackson and Fahrig 2012; Miguet et al. 2016; Moll et al. 2020) to better understand the dynamics of these species and communities. Additionally, long-term monitoring is essential to track temporal changes in response to ongoing urban development including the role of specific environmental factors (e.g., noise and light pollution) in shaping mammal communities and activity patterns.

One of the main challenges for maintaining ecological connectivity of mammals in urban areas is the presence of physical barriers, such as roads, buildings, and paved areas (Peng et al. 2017). These barriers fragment habitats and make it difficult or even prevent the movement of mammals between different areas, regardless of the distance between vegetation patches. Several conservation strategies have been proposed to tackle this issue of declining connectivity. One of them is the establishment of urban ecological corridors (Wang et al. 2022). These corridors are vegetated pathways within urban areas, such as green corridors, pollinator gardens, or tunnels, designed to facilitate the movement of mammals between different sites (Magioli et al. 2016). These green area patches act as crucial stepping-stones, connecting fragmented habitats and facilitating the movement of species across the urban environments (Wang et al. 2022).

The spatial arrangement of these green patches contributes to the formation of a network that mitigates the isolation of natural habitats, fostering resilience in the face of urbanization pressures. It is therefore crucial to conserve the four main patches and the potential corridors, identified here as significantly important to connectivity across the city (Fig. 2). Implementing such measures not only supports the maintenance of functional diversity, but also safeguards genetic diversity and the overall health of mammal populations within urban regions. The significance of these patches extends beyond ecological connectivity for mammals, providing vital ecosystem services such as air purification, temperature regulation, and water retention, thereby enhancing the overall quality of urban life (e.g., Yu et al. 2012; Lepczyk et al. 2017; Zambrano et al. 20). Therefore, by prioritizing the conservation and strategic expansion of green corridors, we can promote the persistence of wildlife within CDMX. We recommended that significant patches and corridors identified here (e.g., such as urban gardens, pollinator gardens, green roofs, and urban parks) be integrated into the Special Green Infrastructure Program of Mexico City (PEIV-Ciudad de México), which is a government initiative coordinated by the Secretariat of the Environment (see <https://sedema.cdmx.gob.mx/programas/programa/infraestructura-verde#>).

We acknowledge important limitations in our study. Firstly, we mixed data obtained by different sampling techniques and degrees of sampling effort to describe the species richness patterns and assemblages of mammals. Each sampling method allows the detection of some, but not all, of the species. Indeed, each type of data involves uncertainty sources (e.g., uneven sample efforts or spatial biases) that cannot be fully eliminated from our results and can have an impact on the observed patterns (Zwerts et al. 2021). Secondly, the diversity estimates were only based on presence/absence of species, but this approach have been fairly criticized for not accounting for differences in detection probability for both spatio-temporal scales, and for ignoring the abundance or density patterns of species (e.g., Sollmann et al. 2013; Parsons et al. 2017).

Additionally, we only considered abiotic effects that impact functional connectivity, but ecological interactions (such as interspecific competition) also represent further challenges for many species. Thus, further research is needed to explore the importance of these interactions in urban environments, where exotic species tend to be common. Finally, although using mammals as a focal group is a first (valuable) step to advance the conservation agenda and urban planning, more research evaluating other taxa is needed to guide effective decisions about ecological connectivity across the CDMX landscape. Despite these limitations, our study show how multiple methods can be used to

answer pressing ecological questions and provide improved guidance for conservation policy in the absence of more rigorous data. In the context of the transdisciplinary research needed for effective conservation and urban planning, we hope to motivate biologists to delve more deeply into analysis of species diversity and develop fast but rigorous and integrative methods and approaches capable to inform conservation and development decisions.

In summary, our study underscores that mammal communities and resilience of ecosystems may benefit from strategies focused on safeguarding, expanding and connecting the green spaces in both urban and peri-urban areas in CDMX. It is vital to educate the public about how they can contribute to preserving biota and natural areas within cities through sustainable practices (e.g. gardens to improve animal movement) and actions (e.g., avoiding the use of poisons). In fact, we argue that increasing participation in citizen science may empower people to take action, as well as offering opportunities for public outreach via citizen science ambassadors, garnering interest in urban ecology, urban wildlife and conservation. Definitively, collaborative efforts between city planners, environmental scientists, and local communities are essential for developing and implementing effective strategies to balance urban development with ecological conservation.

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**Author contributions** PCHR, MCA and DAP-T conceived and designed the study. PCHR, CM, DMR, FB, and DAPT compiled the database of available records and performed the fieldwork. PCHR, JJVTU and DAPT performed the spatial and statistical analyses. All authors contributed equally to the analysis and interpretation of results and writing of the manuscript.

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**Data availability** No datasets were generated or analysed during the current study.

## Declarations

**Competing interests** The authors declare no competing interests.

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