





Research Article

Vegetation changes along an urbanisation and atmospheric pollution gradient in Mexico

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Abstract

Green areas are important places for biodiversity conservation within cities, but their vegetation is affected by various anthropogenic factors. This study used an exploratory approach to examine the influence of urbanisation and air pollution-related factors on the indicators for the composition and structure of vegetation in an urban area in northeast Mexico. Based on the spatial analysis of the major air pollutants, four sampling categories were delimited (rural, low, moderate and high urbanisation). The differences between categories, based on vegetation structure, were determined using non-parametric Kruskal-Wallis tests. The Importance Value was calculated for the species. The floristic similarity was compared using NMDS and PERMANOVA unidirectional. The relationship between environmental variables and abundance of species was evaluated using CCA. One hundred and ten plant species were collected, including ten alien species. The highest abundance and species richness were registered in the rural site. The general tendency of vegetation structure is to plants decreasing with respect to the increase in the levels of urbanisation and air pollution present in the study area. The association between the environmental variables and plant communities along the urbanisation gradient was significant, being the relative humidity, the particles lower than 2.5 μm , the dew point and the heat index as the most important variables. The understanding of the nature and variability of vegetation within green areas contributes to increasing our knowledge about the distribution of the environmental services they provide and the composition of the faunal communities that depend on them. For this reason, this study relates the plants of a specific area of northeast Mexico with the environmental quality present in an urban area.

Key words: air pollution, environmental variability, Monterrey Metropolitan Area, urbanisation, vegetation structure

Introduction

The demographic growth dynamics faced by cities represent a serious threat to the environment, as well as to the health and quality of life of its inhabitants (Vlahov and Galea 2002). The unsustainable use of natural resources, intense land-use changes, increasing density of urban/industrial centres and the



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growing emission of pollutants irreversibly damage the environment (García et al. 2013). These effects not only harm living beings, but also generate phenomena that affect the ecosystem (López et al. 2001). Likewise, the accelerated urbanisation changes the structure of cities and affects their climate and that of their surrounding area (Tang et al. 2008). This urbanisation process occurs more rapidly in countries located in regions classified as developing economies. Particularly in Latin America, where it is estimated that 75% of the population live in cities (UN-HABITAT 2010).

In Mexico, air pollution has deteriorated air quality in various cities, including the Metropolitan Area of the Valley of Mexico, the Metropolitan Area of Guadalajara and the Monterrey Metropolitan Area (MMA) (García et al. 2012; Cerón et al. 2014; Mancilla et al. 2015; Menchaca et al. 2015). It is appropriate to point out that there is a perception problem in society as there is no clear awareness of pollutant emissions, their concentrations and damage to health, urban infrastructure and ecosystems (Lezama and Graizbord 2010). The State of Nuevo León, in the northeast of Mexico, has an unregulated urban growth. Its main urban sprawl, the MMA presents serious environmental problems: geological and hydrological risks, water scarcity, loss of green areas, air pollution, amongst many others (Badillo et al. 2015; Orta et al. 2016; Sanchez-Castillo et al. 2016; Sisto et al. 2016; Ybáñez and Barboza 2017).

Studies of species diversity in urban ecosystems are needed to understand the effect of anthropogenic development on ecosystem integrity and sustenance (Mukherjee et al. 2015). To study the effects of urbanisation on ecosystem structure and function, researchers have used the urban-rural gradient methodology (Pennington et al. 2010). Urban-rural gradients are generally realised on large spatial scales and, in some cases, have been conceived as a linear transect radiating from the city centre towards less disturbed landscapes. Studies employing this method have documented declines in plant species diversity, basal area and density of native species as sites become more urbanised. These studies, which show a decrease in species richness as urbanisation increases, follow a general disturbance hypothesis (Porter et al. 2001; Moffatt et al. 2004; Burton et al. 2005; Duguay et al. 2007).

On the other hand, the intermediate disturbance hypothesis has been one of the main models used to interpret urban plant diversity patterns (Johnson and Swan 2014). The theory has been applied to explore the co-existence of native and non-native species along urban-rural gradients or within the urban environment between patches that vary in level of disturbance (e.g. Porter et al. (2001); ManSecak and Wein (2006); Catford et al. (2012)). The expectation is that species diversity will be maximised in intermediate locations, where native and invasive species are found in the same communities, in relatively uniform proportions.

Previous studies of large-scale urban-rural gradients have documented that those urban forests are more deteriorated than their “natural” or rural counterparts (Paul and Meyer 2001; Güler 2020). Consequently, they reduce the perceived ecological value of remnant vegetation within highly modified landscapes. However, it is important to understand the potential ecological and social value of remnant urban vegetation (Turner et al. 2004; Czaja et al. 2020). Given that more than 60% of the world’s population will reside in urban areas by 2050, these forest fragments in urban settings could provide critical ecosystem

services for both people and other species (Bernhardt and Palmer 2007; Zeg-eye et al. 2023).

For our study, we characterised the remnant vegetation of the MMA, north-east Mexico, along an urbanisation gradient, based on parameters of atmospheric pollution. The objectives of this study were: (1) Identify the plant species richness in the MMA, northeast Mexico; (2) Compare the variation in richness, abundance and diversity of plant species amongst urbanisation categories; (3) Quantify the value of importance of the species by urbanisation category; and (4) Analyse the influence of environmental variation (air pollutants, climatic factors and soil) on the abundance and richness of plant species. Our hypothesis is that the structure and composition of the vegetation decrease with respect to the increase in urbanisation levels in the MMA.

Methods

Study area

The MMA is the largest urban area in northeast Mexico and the third largest urban centre in the country, extending from 25°15' to 26°30' north latitude and from 99°40' to 101°10' west longitude (Fig. 1A, B). The area is bounded by the coastal plain of the Gulf of Mexico and the Sierra Madre Oriental Mountain Range. Several municipalities compose the geographical area of MMA: Apodaca, Cadereyta, García, General Escobedo, Guadalupe, Jiménez, Juárez, Monterrey, Salinas Victoria, San Nicolás de los Garza, San Pedro Garza García, Santa Catarina and Santiago (Alanís 2005; González et al. 2011; Mancilla et al. 2015). The main vegetation cover found at MMA is forest, scrubs and grasslands (Carpio et al. 2021). The MMA has a vehicle fleet of 2.5 million vehicles (Castillo-Nava et al. 2020) and 5.3 million inhabitants (INEGI 2021), which is probably even higher today. Likewise, there is a variety of industrial complexes that include the production of glass, steel, cement and paper, amongst others (Menchaca et al. 2015). The city centre has an average altitude of 540 m a.s.l., the characteristic climate is dry steppe, hot and extreme with temperatures above 35 °C during the summer and below 8 °C during the winter (Alanís 2005; González et al. 2011; Menchaca et al. 2015).

Delimitation of the urbanisation gradient

Since November 1992, the MMA has operated a network of air quality monitoring stations known as the Integral Environmental Monitoring System (SIMA). The SIMA network is currently made up of 14 recording stations distributed according to criteria from meteorological, land use and population density studies. The measurements recorded at these monitoring stations are: PM₁₀ (particulate matter less than 10 µm), PM_{2.5} (particulate matter less than 2.5 µm), carbon monoxide (CO), ozone (O₃), nitrogen oxides (NO_x) and sulphur dioxide (SO₂). In addition, some meteorological variables are reported, such as barometric pressure, rainfall, relative humidity, solar radiation, temperature and wind direction and magnitude (Arreola and González 1999; González et al. 2011; Mancilla et al. 2015). The data recorded by the SIMA stations for air quality and meteorological variables (2009–2018) were obtained from the National

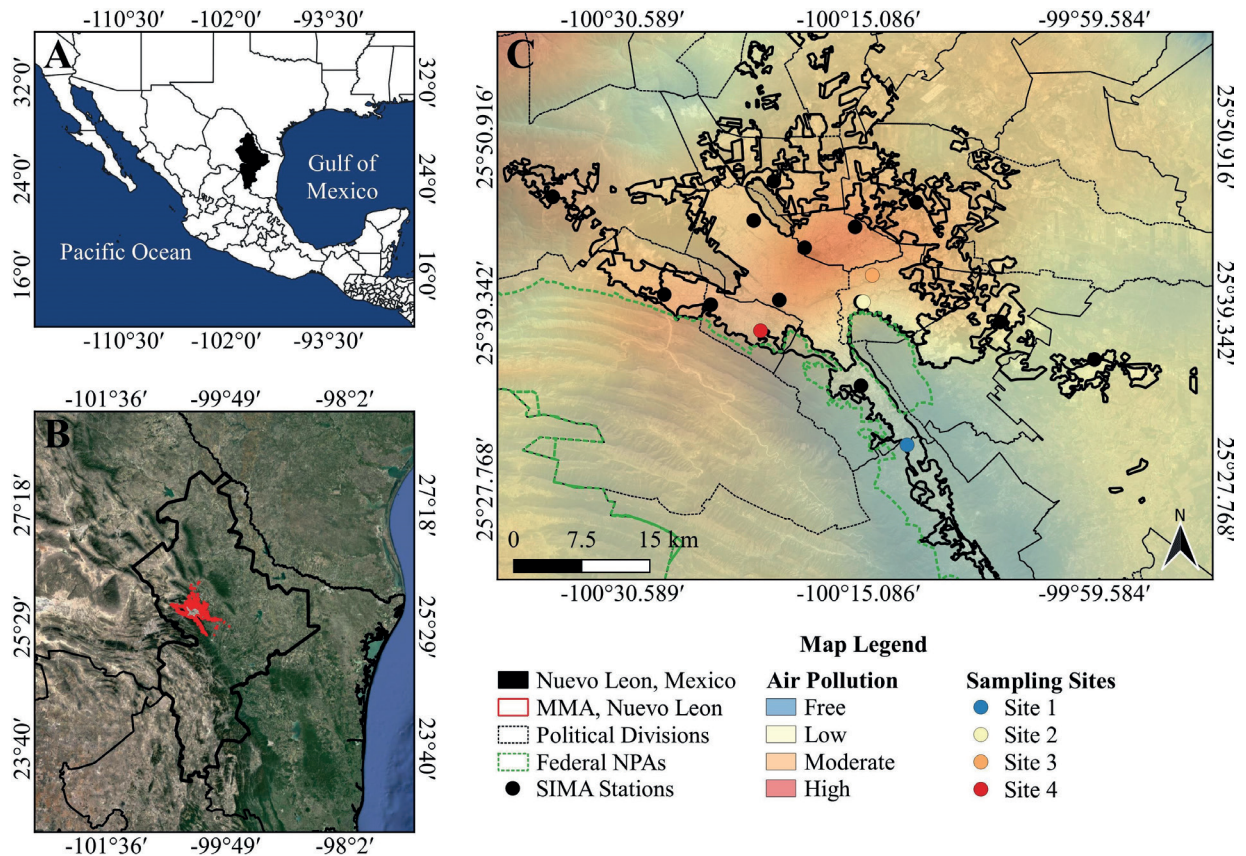


Figure 1. Study area and location of sampling sites **A** location of Nuevo Leon in Mexico **B** location of the MMA inside Nuevo Leon **C** location of sampling sites according with the air pollution levels.

Air Quality Information System (SINAICA). Obtaining descriptive measures for each year and for each of the recording stations was carried out in the Statistica 13.3 programme (TIBCO Software Inc. 2017).

To identify the main pollutants that describe air quality in the MMA during the 2009–2018 period, a Principal Component Analysis (PCA) was carried out. Subsequently, to differentiate the changes in the spatial distribution of pollutants that are indicators of air quality in the MMA, maps were created using the annual average information on each monitoring station. Mapping was done using Inverse Distance Weighting (IDW) interpolation, with a Distance Coefficient of 2 and the output raster pixel size reset to 15 metres. As a reference of the extension, the minimum and maximum distances of the vector sections corresponding to the urban areas that make up the MMA were taken; these sections were obtained from the national layer of Land Use and Vegetation Series 6 (INEGI 2016). The procedures described above were performed using Quantum GIS 3.2 software (Quantum GIS Development Team 2018). As a result, four categories of urbanisation by atmospheric pollution were generated: rural (lower than $3.22 \mu\text{g}/\text{m}^3$ of $\text{PM}_{2.5}$), low (3.22 to $10.56 \mu\text{g}/\text{m}^3$ of $\text{PM}_{2.5}$), moderate (10.56 to $17.92 \mu\text{g}/\text{m}^3$ of $\text{PM}_{2.5}$) and high (17.92 to $25.3 \mu\text{g}/\text{m}^3$ of $\text{PM}_{2.5}$) (Fig. 1C).

Selection of sampling sites

Four permanent sampling sites were delimited, based on the spatial superposition of three geographic elements: (1) the interpolation of the main air pollutants

was used for determination of the urbanisation gradient in the study area (Fig. 1C); (2) images obtained from the Google Earth Pro software were used to differentiate the spatial presence or absence of vegetation cover and (3) a mesh with a grid size of 150 × 150 m was delimited to select sampling areas with complete vegetation cover. Overlay and selection procedures were performed in Quantum GIS 3.2 software. The rural site is located in the Municipality of Santiago, a rural area without substantial urbanisation or air pollution and with secondary submontane scrub vegetation (25°30'41.184"N, 100°11'53.159"W). The low urbanisation site is located in the central zone of the Municipality of Guadalupe with low values of air pollution and secondary vegetation of submontane scrub (25°40'4.944"N, 100°14'45.564"W). The moderate urbanisation site is located in the northern zone of the Municipality of Guadalupe with moderate air pollution and secondary vegetation of submontane scrub (25°42'44.017"N, 100°13'58.825"W). The high urbanisation site is in the Municipality of San Pedro Garza García with high air pollution and anthropogenic submontane scrub vegetation (25°38'11.112"N, 100°21'30.815"W) (Fig. 1C).

Sample collection and processing

During April 2019, the analysis of preliminary samples obtained in the study area was carried out. The Clench model was used to calculate the minimum sample size to be used, based on the method and parameters indicated by Jiménez-Valverde and Hortal (2003). According to the analysis, between 5 to 8 sampling units are needed to register the 95% of the richness of each site. Vegetation was assessed using 20 quadrats of 10 × 10 m (0.01 hectares), which were evenly distributed amongst the four categories of urbanisation by air pollution (five quadrats per category). The quadrats were located from inside the sampling site (150 × 150 m) (2.25 hectares) randomly and they were placed using the tool random points inside a polygon in Quantum GIS 3.2 software. The quadrats are located in patches of natural and native vegetation. The evaluation was carried once per season: dry season (November to April) and rainy season (May to October), during the period from May 2019 to April 2020. The seasons were defined, based on the historical data of the monthly total values of temperature and rainfall (average from 2009 to 2018), which were obtained from the SIMA stations located within the study area (Fig. 2).

Measurements were carried out independently for each of the vegetation strata. For the herbaceous stratum, five sub-quadrats of 1 × 1 m (5 m² in total per quadrant) were delimited. In the shrub stratum, two 5 × 5 m sub-quadrats (50 m² in total per quadrant) were evaluated. Finally, the tree stratum was evaluated in the entire quadrat, 10 × 10 m (100 m² in total per quadrat). The dimension of the quadrat and sub-quadrats was established according to the criteria described by Brower et al. (1998).

In each quadrat/sub-quadrats, the following measurements were made: (1) height of the plant (from its base at ground level to the highest branch); (2) largest and (3) smallest diameter of the aerial projection of the plant. The number of individuals of each morphospecies assigned in the field was quantified and their identification in the laboratory was carried out using the works of Alanís and González (2003), Stubbendieck et al. (2003) and Zurita and Elizondo (2009); likewise, the botanical nomenclature was homogenised using

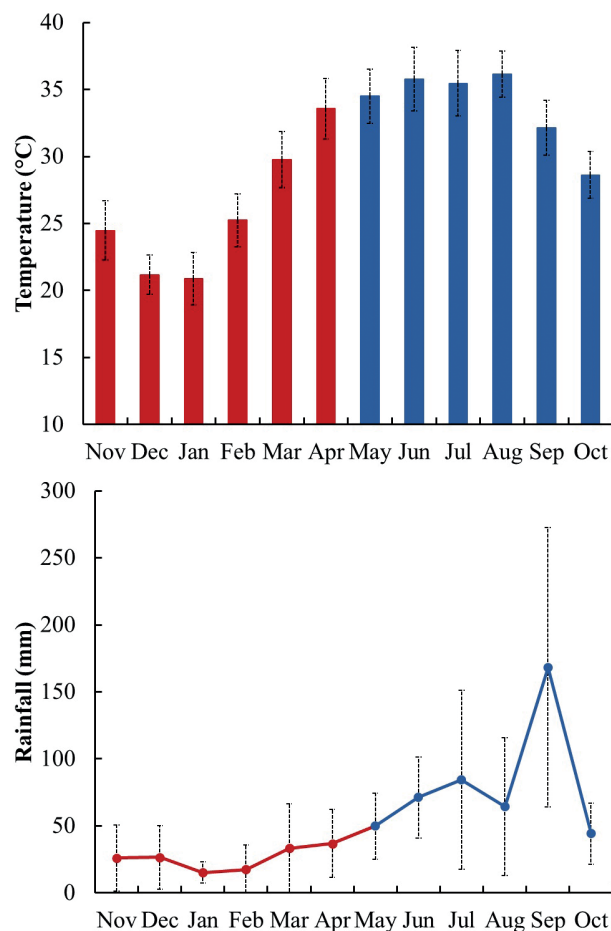


Figure 2. Monthly average variation of temperature and rainfall in the MMA. Dry season (red colour) and Rainy season (blue colour).

the International Plant Names Index base (IPNI 2022). Villaseñor and Espinosa-García (2004) were mainly followed to determine which plant species were not native to the MMA region.

Microenvironment measurement

The microenvironmental variables were measured in each of the quadrants using a Kestrel 5500 portable weather station, a CEM – DT1308 digital luxmeter, a CEM – DT9881 particle counter and a HB – 2 soil moisture and pH meter, simultaneously with the sampling of the vegetation, recording the following variables: maximum wind speed (MWS) and average wind speed (AWS) (obtained during five minutes of exposure), temperature (T), relative humidity (RH), heat index (HI), dew point (DP), evapotranspiration (E), solar radiation (SR), particles of 2.5 ($PM_{2.5}$) and 10 microns (PM_{10}), soil pH (SpH) and soil moisture (SM). Measurements were carried out in the centre of each quadrat during the early hours of the morning, noon and before sunset, avoiding direct solar radiation.

Data analysis

Species richness was measured as the total number of species observed in the study area, as well as in each of the sites. Significant differences in the

number of species between sites were determined using non-parametric Kruskal-Wallis tests, in Statistica 13.3 software. Sampling efficiency was calculated for the entire study area and for each site using the interpolation and extrapolation methodology proposed by Chao and Jost (2012), available in the iNEXT package (Hsieh et al. 2016) for version 3.5.3 of R (R Development Core Team 2019).

Differences in plant abundance between sites were calculated with a Kruskal-Wallis test. For the analysis of alpha diversity, we adopted the analytical method of Chao and Jost (2015) to obtain profiles in which diversity is evaluated in terms of “effective numbers of species” (qD), an approach that is equivalent to the numbers of Hill (Hill 1973). Hill numbers include three widely-used measures as special cases: species richness ($q = 0$), Shannon diversity (the exponential of Shannon entropy, $q = 1$) and Simpson diversity (the inverse of Simpson concentration, $q = 2$), all of which are expressed in units of “species equivalents”. The analysis was performed for the entire study area and for each site using the SpadeR package (Chao et al. 2016), in R 3.5.3.

Vegetation cover was calculated according to the criteria described by Ramírez (2006). Differences in vegetation cover between sites were determined with a Kruskal-Wallis test. To examine differences in species composition between sites, we performed non-metric multidimensional scaling (NMDS) analysis, using the Bray-Curtis Index as the similarity matrix. A PERMANOVA was also performed to test for differences in species composition between sites. Both analyses were performed using the Vegan package (Oksanen et al. 2019) in R 3.5.3.

For each species, its abundance was determined according to the number of individuals, its dominance based on cover and its frequency based on its presence in the sampling quadrats. These results were used to obtain a weighted value at the taxon level called Importance Value (IV), which acquires percentage values on a scale from 0 to 100 (Mueller-Dombois and Ellenberg 1974). The IV was calculated for each site separately.

Finally, a canonical correspondence analysis (CCA) was carried out to determine the relationship between the microenvironmental variables and the abundance of the recorded species in each plot, which also includes a Monte Carlo permutation test to evaluate the significance of the microenvironmental variables in the analysis. For the CCA, the average values of the microenvironmental variables of each season of the year were used (dry and rainy season). The CCA was done using the Vegan package in R 3.5.3.

Results

A total of 12,878 plants of 42 families, 104 genera and 110 species were quantified. From this total, 17 species (594 individuals) were trees, 34 (2,595 individuals) were shrubs and 59 (9,689 individuals) were herbaceous (Table 1). The greatest abundance and richness of tree species in the study area was found in the Fabaceae family with 35.0 and 23.5% of the total registered, respectively. Likewise, Fabaceae presented the highest abundance and richness of shrub species with 29.3 and 29.4% of the total registered, respectively. Asteraceae showed the highest abundance and richness of herbaceous species with 21.6 and 18.6% of the total recorded, respectively (Table 1).

Table 1. Taxonomic list, abundance and IV of the species found in an air pollution gradient in the MMA. Legend: Site 1 = Rural, Site 2 = Low urbanisation, Site 3 = Moderate urbanisation, Site 4 = High urbanisation.

Taxon	Key	Abundance				IV			
		Site 1	Site 2	Site 3	Site 4	Site 1	Site 2	Site 3	Site 4
Tree									
Boraginaceae Juss.									
<i>Ehretia anacua</i> (Terán & Berland.) I.M. Johnst.	Eana	16	6	0	0	8.1	4.7	0.0	0.0
Cannabaceae Martinov									
<i>Celtis laevigata</i> Willd.	Clae	26	22	0	0	12.1	12.7	0.0	0.0
Ebenaceae Gürke									
<i>Diospyros texana</i> Scheele	Dtex	22	0	0	0	10.7	0.0	0.0	0.0
Fabaceae Lindl.									
<i>Ebenopsis ebano</i> (Berland.) Barneby & J.W. Grimes	Eeba	20	28	16	0	10.2	14.6	17.3	0.0
<i>Havardia pallens</i> (Benth.) Britton & Rose	Hpal	18	12	6	0	9.6	8.7	5.7	0.0
<i>Leucaena leucocephala</i> (Lam.) de Wit*	Lleu	0	26	16	0	0.0	12.6	18.9	0.0
<i>Prosopis glandulosa</i> Torr.	Pgla	26	18	22	0	12.4	9.6	21.2	0.0
Fagaceae Dumort.									
<i>Quercus fusiformis</i> Small	Qfus	0	0	0	24	0.0	0.0	0.0	22.4
Juglandaceae DC. ex Perleb									
<i>Carya illinoensis</i> (Wangenh.) K. Koch	Cill	0	0	0	6	0.0	0.0	0.0	5.1
Oleaceae Hoffmanns. & Link									
<i>Fraxinus americana</i> L.*	Fame	0	0	14	16	0.0	0.0	16.0	16.9
<i>Ligustrum lucidum</i> W.T. Aiton*	Lluc	0	0	0	22	0.0	0.0	0.0	18.0
Rutaceae Juss.									
<i>Sargentia greggii</i> S. Watson	Sgre	20	0	0	0	9.3	0.0	0.0	0.0
Salicaceae Mirb.									
<i>Salix nigra</i> Marshall	Snig	0	22	0	0	0.0	11.8	0.0	0.0
Sapindaceae Juss.									
<i>Koelreuteria elegans</i> (Seem.) A.C. Sm.*	Kele	0	30	22	16	0.0	14.1	21.0	14.1
<i>Sapindus saponaria</i> L.	Ssap	34	0	0	0	15.4	0.0	0.0	0.0
<i>Ungnadia speciosa</i> Endl.	Uspe	0	0	0	26	0.0	0.0	0.0	23.5
Sapotaceae Juss.									
<i>Sideroxylon celastrinum</i> (Kunth) T.D. Penn.	Scel	24	18	0	0	12.2	11.2	0.0	0.0
Shrub									
Asparagaceae Juss.									
<i>Yucca treculeana</i> Carrière	Ytre	33	0	0	0	2.9	0.0	0.0	0.0
Asteraceae Bercht. & J. Presl									
<i>Gochnatia hypoleuca</i> (DC.) A. Gray	Ghyp	34	0	0	0	3.0	0.0	0.0	0.0
Bignoniaceae Juss.									
<i>Tecoma stans</i> (L.) Juss. ex Kunth	Tsta	0	59	0	0	0.0	7.5	0.0	0.0
Boraginaceae Juss.									
<i>Cordia boissieri</i> A. DC.	Cboi	54	44	0	0	4.4	6.3	0.0	0.0
Cactaceae Juss.									
<i>Opuntia engelmannii</i> Salm-Dyck ex Engelm.	Oeng	40	0	0	0	3.6	0.0	0.0	0.0

Taxon	Key	Abundance				IV			
		Site 1	Site 2	Site 3	Site 4	Site 1	Site 2	Site 3	Site 4
Cannabaceae Martinov									
<i>Celtis pallida</i> Torr.	Cpal	53	32	0	0	4.2	4.9	0.0	0.0
Capparaceae Juss.									
<i>Capparis flexuosa</i> Vell.	Cfle	39	0	0	0	3.4	0.0	0.0	0.0
Euphorbiaceae Juss.									
<i>Adelia vaseyi</i> (JM Coult.) Pax y K. Hoffm.	Avas	43	36	0	0	3.7	5.4	0.0	0.0
Fabaceae Lindl.									
<i>Acacia berlandieri</i> Benth.	Aber	59	40	0	0	4.7	5.9	0.0	0.0
<i>Acacia farnesiana</i> (L.) Willd.	Afar	0	0	53	0	0.0	0.0	12.8	0.0
<i>Acacia rigidula</i> Benth.	Arig	49	0	0	0	4.1	0.0	0.0	0.0
<i>Bauhinia mexicana</i> Vogel	Bmex	36	0	0	0	3.4	0.0	0.0	0.0
<i>Caesalpinia mexicana</i> A. Gray	Cmex	0	33	50	61	0.0	5.0	12.6	19.6
<i>Dalea scandens</i> (Mill.) R.T. Clausen	Dsca	45	0	0	0	3.9	0.0	0.0	0.0
<i>Erythrina herbacea</i> L.	Eher	43	0	0	0	3.8	0.0	0.0	0.0
<i>Eysenhardtia texana</i> Scheele	Etex	28	38	40	45	3.0	5.5	11.0	15.2
<i>Mimosa monancistra</i> Benth.	Mmon	0	40	43	0	0.0	5.8	11.7	0.0
<i>Parkinsonia aculeata</i> L.	Pacu	0	0	57	0	0.0	0.0	14.2	0.0
Lythraceae J. St.-Hil.									
<i>Punica granatum</i> L.*	Pgra	0	0	0	41	0.0	0.0	0.0	15.1
Malpighiaceae Juss.									
<i>Malpighia glabra</i> L.	Mgla	46	33	0	0	3.9	5.1	0.0	0.0
<i>Mascagnia macroptera</i> (Moc. & Sessé ex DC.) Nied.	Mmac	54	50	48	63	4.3	6.7	12.3	19.6
Myrtaceae Juss.									
<i>Psidium guajava</i> L.*	Pgua	0	0	0	41	0.0	0.0	0.0	14.6
Oleaceae Hoffmanns. & Link									
<i>Forestiera angustifolia</i> Torr.	Fang	29	45	0	0	2.8	6.2	0.0	0.0
Rhamnaceae Juss.									
<i>Condalia hookeri</i> M.C. Johnst.	Choo	49	56	0	0	4.0	7.3	0.0	0.0
<i>Karwinskia humboldtiana</i> (Schult.) Zucc.	Khum	51	0	0	0	4.3	0.0	0.0	0.0
<i>Ziziphus obtusifolia</i> (Hook. ex Torr. & A. Gray) A. Gray	Zobt	33	0	0	0	2.9	0.0	0.0	0.0
Rubiaceae Juss.									
<i>Randia obcordata</i> S. Watson	Robc	42	53	0	0	3.6	7.2	0.0	0.0
Rutaceae Juss.									
<i>Helietta parvifolia</i> (A. Gray ex Hemsl.) Benth.	Hpar	45	41	48	0	3.8	5.7	12.5	0.0
<i>Zanthoxylum fagara</i> (L.) Sarg.	Zfag	47	41	0	0	3.8	5.9	0.0	0.0
Salicaceae Mirb.									
<i>Neopringlea integrifolia</i> (Hemsl.) S. Watson	Nint	58	0	0	0	4.5	0.0	0.0	0.0
Scrophulariaceae Juss.									
<i>Leucophyllum frutescens</i> (Berland.) I.M. Johnst.	Lfru	35	36	0	0	2.9	5.4	0.0	0.0
Simaroubaceae DC.									
<i>Castela erecta</i> Turpin	Cere	49	0	0	0	3.5	0.0	0.0	0.0
Verbenaceae J. St.-Hil.									
<i>Citharexylum berlandieri</i> B.L. Rob.	Cber	43	0	0	0	3.2	0.0	0.0	0.0

Taxon	Key	Abundance				IV			
		Site 1	Site 2	Site 3	Site 4	Site 1	Site 2	Site 3	Site 4
<i>Lantana camara</i> L.	Lcam	66	26	52	47	4.5	4.1	12.9	15.9
Herb									
Acanthaceae Juss.									
<i>Elytraria bromoides</i> Oerst.	Ebro	79	0	0	0	3.0	0.0	0.0	0.0
<i>Justicia pilosella</i> (Nees) Hilsenb.	Jpil	80	114	0	0	3.0	4.2	0.0	0.0
<i>Ruellia nudiflora</i> (Engelm. & A. Gray) Urb.	Rnud	0	122	0	0	0.0	4.3	0.0	0.0
<i>Tetramerium nervosum</i> Nees	Tner	101	0	0	0	3.5	0.0	0.0	0.0
Apocynaceae Juss.									
<i>Asclepias curassavica</i> L.	Acur	101	76	96	90	3.5	3.2	4.0	5.5
<i>Telosiphonia lanuginosa</i> (M. Martens & Galeotti) Henrickson	Tlan	0	0	105	104	0.0	0.0	4.3	5.9
Asteraceae Bercht. & J. Presl									
<i>Bidens odorata</i> Cav.	Bodo	114	94	119	114	3.8	3.7	4.5	6.3
<i>Calyptocarpus vialis</i> Less.	Cvia	0	101	103	93	0.0	3.8	4.1	5.7
<i>Chromolaena odorata</i> (L.) R.M. King & H. Rob.	Codo	111	0	0	0	3.8	0.0	0.0	0.0
<i>Helianthus annuus</i> L.	Hann	0	0	94	0	0.0	0.0	3.9	0.0
<i>Jefea lantanifolia</i> (S. Schauer) Strother	Jlan	0	93	0	0	0.0	3.6	0.0	0.0
<i>Sanvitalia ocymoides</i> DC.	Socy	64	97	0	0	2.1	3.8	0.0	0.0
<i>Simsia eurylepis</i> S.F. Blake	Seur	0	77	95	0	0.0	3.3	4.0	0.0
<i>Thymophylla pentachaeta</i> (DC.) Small	Tpen	0	0	0	132	0.0	0.0	0.0	7.1
<i>Tridax coronopifolia</i> (Kunth) Hemsl.*	Tcor	99	101	103	131	3.4	3.8	4.2	6.8
<i>Verbesina persicifolia</i> DC.	Vper	61	0	0	0	2.1	0.0	0.0	0.0
<i>Wedelia acapulcensis</i> Kunth	Waca	96	0	0	0	3.5	0.0	0.0	0.0
Commelinaceae Mirb.									
<i>Commelina erecta</i> L.	Cere	99	89	95	87	3.3	3.5	3.9	5.3
Convolvulaceae Juss.									
<i>Evolvulus alsinoides</i> (L.) L.	Eals	0	0	119	0	0.0	0.0	4.6	0.0
<i>Ipomoea hederacea</i> Jacq.	Ihed	91	84	69	0	3.3	3.3	3.1	0.0
<i>Merremia dissecta</i> (Jacq.) Hallier f.	Mdis	0	101	98	0	0.0	3.7	3.9	0.0
Euphorbiaceae Juss.									
<i>Acalypha monostachya</i> Cav.	Amon	0	0	0	82	0.0	0.0	0.0	5.1
<i>Cnidioscolus rotundifolius</i> (Müll. Arg.) McVaugh	Crot	0	103	0	0	0.0	3.9	0.0	0.0
<i>Croton cortesianus</i> Kunth	Ccor	103	114	0	0	3.6	4.3	0.0	0.0
<i>Euphorbia hirta</i> L.	Ehir	0	78	109	0	0.0	3.3	4.3	0.0
Fabaceae Lindl.									
<i>Canavalia villosa</i> Benth.	Cvil	101	0	0	0	3.5	0.0	0.0	0.0
<i>Desmanthus virgatus</i> (L.) Willd.	Dvir	95	0	0	0	3.4	0.0	0.0	0.0
<i>Mimosa malacophylla</i> A. Gray	Mmal	112	0	0	0	3.7	0.0	0.0	0.0
Lamiaceae Martinov									
<i>Ocimum micranthum</i> Willd.*	Omic	81	76	0	0	3.1	3.2	0.0	0.0
<i>Salvia coccinea</i> Buc'hoz ex Etl.	Scoc	93	102	106	105	3.4	3.9	4.1	5.9
Loasaceae Juss.									
<i>Cevallia sinuata</i> Lag.	Csin	0	0	0	89	0.0	0.0	0.0	5.3

Taxon	Key	Abundance				IV			
		Site 1	Site 2	Site 3	Site 4	Site 1	Site 2	Site 3	Site 4
Malvaceae Juss.									
<i>Abutilon trisulcatum</i> (Jacq.) Urb.	Atri	0	105	113	0	0.0	3.9	4.3	0.0
<i>Malvastrum americanum</i> (L.) Torr.	Mame	99	0	0	0	3.4	0.0	0.0	0.0
<i>Melochia pyramidata</i> L.	Mpyr	78	95	99	96	2.9	3.7	4.0	5.5
<i>Waltheria indica</i> L.	Wind	0	0	97	0	0.0	0.0	4.0	0.0
Nyctaginaceae Juss.									
<i>Cyphomeris crassifolia</i> (Standl.) Standl.	Ccra	57	0	0	0	2.1	0.0	0.0	0.0
Oleaceae Hoffmanns. & Link									
<i>Menodora heterophylla</i> Moric. ex DC.	Mhet	0	0	63	0	0.0	0.0	2.5	0.0
Onagraceae Juss.									
<i>Oenothera rosea</i> L'Hér. ex Aiton	Oros	0	0	55	0	0.0	0.0	2.2	0.0
Papaveraceae Juss.									
<i>Argemone grandiflora</i> Sweet	Agra	0	0	63	0	0.0	0.0	2.4	0.0
Passifloraceae Juss. ex Roussel									
<i>Passiflora foetida</i> L.	Pfoe	69	64	0	0	2.3	2.3	0.0	0.0
Petiveriaceae C. Agardh									
<i>Rivina humilis</i> L.	Rhum	68	60	0	0	2.3	2.3	0.0	0.0
Poaceae Barnhart									
<i>Aristida adscensionis</i> L.	Aads	100	0	0	0	3.5	0.0	0.0	0.0
<i>Bouteloua curtipendula</i> (Michx.) Torr.	Bcur	0	0	0	104	0.0	0.0	0.0	5.9
<i>Cenchrus spinifex</i> Cav.	Cspi	90	114	0	0	3.3	4.2	0.0	0.0
<i>Eragrostis barrelieri</i> Daveau*	Ebar	0	0	84	0	0.0	0.0	3.7	0.0
<i>Melinis repens</i> (Willd.) Zizka*	Mrep	0	0	91	94	0.0	0.0	3.9	5.4
<i>Panicum hallii</i> Vasey	Phal	0	90	0	0	0.0	3.5	0.0	0.0
<i>Paspalum unispicatum</i> (Scribn. & Merr.) Nash	Puni	0	0	69	0	0.0	0.0	2.7	0.0
<i>Setaria leucopila</i> (Scribn. & Merr.) K. Schum.	Sleu	102	107	0	0	3.5	3.9	0.0	0.0
Pteridaceae E.D.M. Kirchn.									
<i>Adiantum tricholepis</i> Fée	Atri	92	0	0	0	3.4	0.0	0.0	0.0
<i>Cheilanthes aemula</i> Maxon	Caem	88	0	0	0	3.1	0.0	0.0	0.0
Ranunculaceae Juss.									
<i>Clematis drummondii</i> Torr. & A. Gray	Cdru	122	98	134	0	4.1	3.8	5.0	0.0
Rubiaceae Juss.									
<i>Spermacoce glabra</i> Michx.	Sgla	45	67	0	0	1.7	2.5	0.0	0.0
Sapindaceae Juss.									
<i>Cardiospermum halicacabum</i> L.	Chal	0	103	102	84	0.0	3.8	4.1	5.2
Solanaceae Juss.									
<i>Solanum elaeagnifolium</i> Cav.	Sela	0	0	95	0	0.0	0.0	4.0	0.0
<i>Solanum triquetrum</i> Cav.	Stri	54	0	0	0	1.9	0.0	0.0	0.0
Verbenaceae J. St.-Hil.									
<i>Lantana canescens</i> Kunth	Lcan	98	90	106	96	3.4	3.5	4.2	5.6
<i>Phyla nodiflora</i> (L.) Greene	Pnod	0	0	0	136	0.0	0.0	0.0	7.1
<i>Verbena canescens</i> Kunth	Vcan	0	0	0	112	0.0	0.0	0.0	6.4
*Introduced species.									

Koelreuteria elegans (Seem.) A.C. Sm. was the species with the highest abundance of trees (68 individuals), 11.4% of the total number of individuals recorded in the study area. On the other hand, *Mascagnia macroptera* (Moc. & Sessé ex DC.) Nied. presented the highest abundance of shrubs (215 individuals) (8.3%). *Bidens odorata* Cav. was the species with the highest abundance of herbaceous plants (441 individuals) (4.6%) (Table 1). From the total of the reported species, ten are alien species, amongst which, *Koelreuteria elegans* (Seem.) A.C. Sm. and *Tridax coronopifolia* (Kunth) Hemsl. are present in most of the sampling sites (Table 1).

Variation per urbanisation category

Sapindus saponaria L. was the tree species with the highest IV (15.4%) in the rural site. On the other hand, *Ebenopsis ebano* (Berland.) Barneby & J.W. Grimes was the most important tree species (14.61%) in the low urbanisation site. *Prosopis glandulosa* Torre. was the most important species (21.21%) in the moderate urbanisation site. Likewise, *Ungnadia speciosa* Endl. was the most important species (23.48%) in the high urbanisation site (Table 1).

Acacia berlandieri Benth. was the shrub species with the highest IV (4.68%) in the rural site. On the other hand, *Tecoma stans* (L.) Juss. ex Kunth was the most important shrub species (7.53%) in the low urbanisation site. *Parkinsonia aculeata* L. was the most important species (14.23%) in the moderate urbanisation site. Likewise, *Caesalpinia mexicana* A. Gray was the most important species (19.63%) in the high urbanisation site (Table 1).

Clematis drummondii Torr. & A. Gray was the herbaceous species with the highest IV (4.06%) in the rural site. On the other hand, *Ruellia nudiflora* (Engelm. & A. Gray) Urb. was the most important herbaceous species (4.31%) in the low urbanisation site. *Clematis drummondii* Torr. & A. Gray also turned out to be the most important herbaceous (5.02%) in the moderate urbanization site. Likewise, *Phyla nodiflora* (L.) Greene turned out to be the most important species (7.12%) in the high urbanisation site (Table 1).

Comparisons between sites showed significant differences ($P < 0.05$) for species richness, height and coverage between all sites (Table 2). Abundance was significantly different ($P < 0.05$) between all sites, except for the comparison between the sites with moderate and high urbanisation (Table 2). All the parameters (abundance, species richness, height and coverage) decreased with increasing levels of urbanisation or pollution. In the rural site, 425.2 ± 87.7 individuals and 61.8 ± 3.4 species were registered, representing a sampling coverage of 99.9%. In the low urbanisation site, the values were reduced to 350 ± 68.5 individuals and 50.9 ± 1.6 species (coverage of 99.9%). For the moderate urbanisation site, 296.9 ± 62.4 individuals and 36.2 ± 1.6 species were registered (coverage of 99.9%), while for the high urbanisation site, 215.7 ± 35.5 individuals and 27.2 ± 0.6 species (coverage of 100%).

For 0D , 1D and 2D , the rural site had the highest diversity. All comparisons between sites were significantly different (with 95% confidence intervals) (Table 2). The one-way PERMANOVA test detected significant differences in species composition between all sites ($SS_{\text{total}} = 7.05$; $SS_{\text{within-group}} = 1.25$; $F = 55.81$, $P < 0.001$). Plant communities sampled formed separate groups in the NMDS diagram (Stress = 0.11) (Fig. 3).

Table 2. Richness, abundance, height, coverage and diversity profiles along the urbanisation gradient in the MMA. Legend: ⁰D = species richness expressed in units of species; ¹D = Shannon diversity expressed in units of species; ²D = Simpson diversity expressed in units of species.

Ecological parameter	Rural	Low urbanisation	Moderate urbanisation	High urbanisation
Richness *	61.8 ± 3.4a	50.9 ± 1.6b	36.2 ± 1.6c	27.2 ± 0.6d
Abundance*	425.2 ± 87.7a	350 ± 68.5b	296.9 ± 62.4c	215.7 ± 35.5c
Height *	1.2 ± 0.1a	0.9 ± 0b	0.7 ± 0c	0.8 ± 0.1d
Coverage *	558.3 ± 103.1a	205.7 ± 31.1b	117.4 ± 22.9c	118.5 ± 19.9d
⁰ D **	68 ± 0a	54 ± 0b	40 ± 0.2c	29 ± 0d
¹ D **	61.2 ± 0.8a	47.3 ± 0.8b	35.5 ± 0.5c	25.1 ± 0.5d
² D **	56.5 ± 1.2a	43.4 ± 1b	33.4 ± 0.7c	23.3 ± 0.7d

* Values with different letters between columns are significantly different using Kruskal-Wallis test: richness between sites, $K = 36.6$, $DF = 3$, $P = 0.0001$; abundance between sites, $K = 17.5$, $DF = 3$, $P = 0.0001$; height between sites, $K = 32.5$, $DF = 3$, $P = 0.0001$; coverage between sites, $K = 31.2$, $DF = 3$, $P = 0.0001$. ** Diversity values with different letters between columns are different, using 95% confidence intervals.

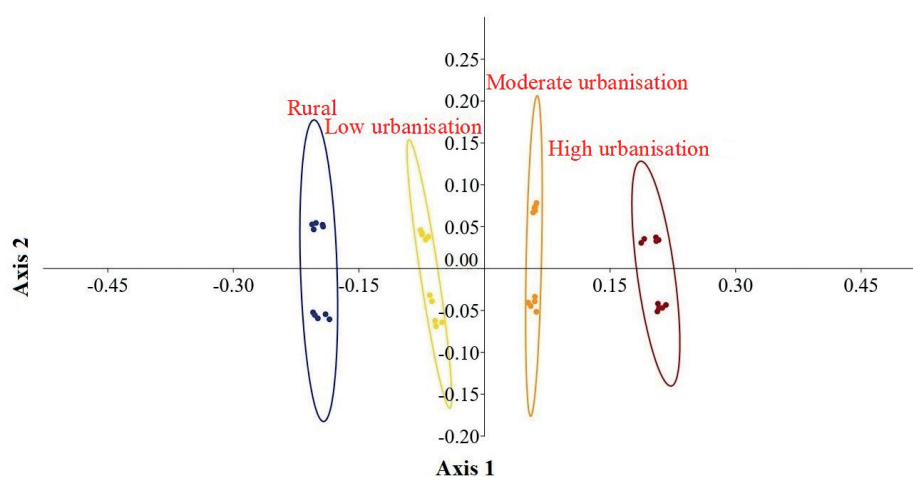


Figure 3. Non-Metric Multidimensional Scaling (NMDS) ordination of plant communities by urbanisation categories.

Plant responses to environmental variation

The MWS, T, RH, HI, DP, E, $PM_{2.5}$ and SpH were the significant environmental variables ($P < 0.05$) used in the CCA (Table 3). CCA showed significant association between the environmental variables and the plant communities along the urbanisation gradient (Total inertia = 81.3%; $P < 0.001$). The variables most related with the plant abundance in the gradient were: RH and $PM_{2.5}$ for Axis 1 (Eigenvalue = 0.441; Inertia = 56.6%). For Axis 2 (Eigenvalue = 0.193; Inertia = 24.7%), DP and HI were the most important variables. *Quercus fusiformis* Small, *Carya illinoensis* (Wangenh.) K. Koch, *Fraxinus americana* L., *Ligustrum lucidum* W.T. Aiton, *Koeleria elegans* (Seem.) A.C. Sm., *Caesalpinia mexicana* A. Gray, *Punica granatum* L., *Psidium guajava* L., *Lantana camara* L., *Thymophylla pentachaeta* (DC.) Small, *Tridax coronopifolia* (Kunth) Hemsl., *Eragrostis barrelieri* Daveau, *Melinis repens* (Willd.) Zizka and *Verbena canescens* Kunth are associated with conditions of high concentration of $PM_{2.5}$, higher RH and alkaline SpH. On the other hand, *Ehretia anacua* (Terán & Berland.) I.M. Johnst., *Diospyros texana* Scheele,

Discussion

We use the air quality records to define an urbanisation gradient in the MMA, where the height, cover, abundance, species richness and diversity were the parameters recorded in sites with different levels of urbanisation. It was found that all the parameters decreased with increasing urbanisation levels; thus, in accordance with the general disturbance hypothesis, the general tendency of plant distribution at the levels of urbanisation and pollution present in the MMA is to decrease.

It is important to note that urban gradient studies are clearly a simplification of the complex patterns produced by urbanisation, such as air pollution (Alberti et al. 2001; Hahs and McDonnell 2006; McKinney 2008). The negative effect of urbanisation on plant species richness has been related to a variety of factors ranging from the pollution and habitat degradation to introduction of alien species and others societal impacts (McKinney 2002, 2006; Hope et al. 2003; Grimm et al. 2008; Vilà et al. 2011). While the impact of urbanisation on plant richness depends upon the size of an urban area, the overall loss of habitable areas in urban zones normally results in a lower richness of plants. On the other hand, the expansion of urban areas is associated with an influx of non-native plant species that tend to counterbalance this urban effect by increasing overall plant richness (Duguay et al. 2007; Gavier-Pizarro et al. 2010; Cameron et al. 2015). These patterns were observed in the MMA region, notably, we recorded a decrease in the number of species as urbanisation levels increase and an increase in the abundance of introduced species in sites with higher urbanisation level.

The integrity of plant communities is vulnerable to intense land-use modification associated with urbanisation (Richardson et al. 2007). Significant changes in species composition along urban-rural gradients have been reported in Baltimore, Maryland (Groffman et al. 2003), Winnipeg, Manitoba (Moffatt et al. 2004) and Columbus, Georgia (Burton et al. 2005; Styers et al. 2010). Species richness and density of native plants were shown to decrease near urban areas (Porter et al. 2001; Moffatt et al. 2004), whereas invasive richness and density increased with urban development in the south-eastern United States. (Burton et al. 2005). These studies applied an urban-to-rural gradient approach to study sites located over a large geographic region from a densely populated urban landscape to a relatively unpopulated rural landscape. Our results corroborate similar studies of declining plant populations in urban-rural gradients, suggesting that habitat degradation may be a devastating threat to the persistence of certain sensitive taxa, such as plants present only in rural sites.

The replacement of local native species by alien species causes the floras of cities in different biogeographic regions to be increasingly homogeneous (i.e. beta diversity is reduced) (Kühn and Klotz 2006; Schwartz et al. 2006). However, the introduction of non-native species in urban areas can make them relatively biologically diverse at smaller scales. Our results show a clear differentiation in species composition (beta diversity) between sites on the urban-rural gradient. Hope et al. (2003) and Turner et al. (2005) demonstrate that certain anthropogenic habitats may have similar or greater alpha diversity than the more natural habitats of the region. However, our results show a greater diversity for sites without apparent urbanisation, but it decreases as urbanisation levels increase. The low diversity in such habitats may reflect a high degree of land change, thus causing significant stress to the plant community in urban areas (Pennington et al. 2010).

Certain native and alien species represent ecological indicators of different levels of urbanisation (LaPaix and Freedman 2010). *Sapindus Saponaria*, *Acacia berlandieri* and *Clematis drummondii* were the species with the highest IV value in the rural site. On the other hand, *Ungnadia speciosa*, *Caesalpinia mexicana* and *Phyla nodiflora* were the most important species in the high urbanisation site. Our species with the highest IV differ from those mentioned by Alanís-Rodríguez et al. (2015) for areas contiguous to the MMA. In contrast, Estrada-Castillón et al. (2012) report plant associations made up of species mentioned in our study, clarifying that the plant communities with the highest deterioration are associated with the areas adjacent to the metropolitan zone.

From the CCA, we identified plant species associated with urbanisation (Kremen 1992). Amongst which, invasive alien species, such as *Ligustrum lucidum*, *Koelreuteria elegans*, *Tridax coronopifolia*, *Eragrostis barrelieri* and *Melinis repens*, were found in the more urbanised sites. These species are highly tolerant to urban growth conditions and appear capable of exploiting environmental conditions associated with urbanisation (McKinney 2002). Native species, such as *Quercus fusiformis*, *Carya illinoensis*, *Caesalpinia Mexicana*, *Lantana camara*, *Thymophylla pentachaeta* and *Verbena canescens*, were amongst the most common species to observe in urbanised sites and likely present adaptations capable of tolerating disturbance associated with urbanisation. In contrast, native species, such as *Diospyros texana*, *Sapindus saponaria*, *Gochnatia hypoleuca*, *Zanthoxylum fagara*, *Citharexylum berlandieri*, *Verbesina persicifolia*, *Solanum triquetrum* and *Lantana canescens*, were found only in the less urbanised sites. Consequently, these species are highly intolerant to process associated with urbanisation, highlighting the importance of green areas as refuges for these species. These results are consistent with the large-scale studies by Moffatt et al. (2004) and Burton and Samuelson (2008), who reported a predominance of exotic and pioneer species in more urbanised areas compared to rural areas.

The composition and structure of vegetation in peri-urban and urban areas can vary due to climate, soil conditions, ecological disturbances and human influences (Jim and Liu 2001; Jim 2002; Pedlowski et al. 2002; Escobedo et al. 2006). For this study, the conditions of RH, DP, HI and $PM_{2.5}$ were the variables that best describe the vegetation structure in the MMA. Other studies have documented these characteristics. For example, Stewart et al. (2009) in New Zealand and Godefroid and Koedam (2003) in Belgium studied different plant assemblages in urban and peri-urban temperate forests. In Latin America, Grau et al. (2008) in Tucumán, Argentina and Baumgardner et al. (2012) in Mexico City, analysed the role of the structure and composition of peri-urban forests as a function of the watershed and regional air quality, respectively. In addition, Puric-Mladenovic et al. (2000) in Canada and Christopoulou et al. (2007) in Greece, discussed the loss of peri-urban natural areas due to urbanisation.

Other anthropogenic factors of vegetation structure and composition have been found in other urban and subtropical areas of the world (Jim 2002; Grau et al. 2008). For example, people in southern China prefer green areas characterised by high tree cover and large trees (Jim and Chen 2006). Furthermore, socioeconomic and educational levels may play a role in the structure and composition of forests in Brazilian urban areas (Pedlowski et al. 2002). In Kenya, peri-urban mangroves have been affected by industrial pollution and sewage (Mohamed et al. 2009).

The approach used in this research implies a relationship between microclimatic variations and plant species at the plot level. This analysis assumes the influence of environmental variables (independent variables) on the species (Dolédec et al. 2000). However, the relationship between both factors is interdependent. That is, the structure of the vegetation and the characteristics of the plants influence the abiotic variation (Guariguata and Ostertag 2001; Renaud et al. 2010; Lienard et al. 2015; Hardwick et al. 2015) and, at the same time, the presence of certain microclimatic conditions allows the development of each plant species (Arroyo-Rodríguez et al. 2017). Therefore, the microclimate is one of the first factors to change after disturbance (Norris et al. 2012; Parr 2012; Hardwick et al. 2015).

Overall, our study analysed the effects of urbanisation on vegetation and changes in vegetation structure were detected as levels of urbanisation increased. However, studies in subtropical regions of North America show how, in addition to urbanisation, demography also affects the structure of the vegetation, mainly tree structure in built-up areas (Zhao et al. 2010; Flocks et al. 2011). Additionally, other studies in South America document the effect of socioeconomics in vegetation structure (De la Maza et al. 2002; Pedlowski et al. 2002; Escobedo et al. 2006). The ability of parks and areas of remaining native vegetation to promote biodiversity depends largely on their design and the types of management activities to which they are subjected. For example, while regionally rare native species can be found within cities, they are often associated with habitats that have not been greatly altered (Godefroid 2001; Godefroid and Koedam 2003). Such partnerships strengthen the call to protect plant communities within the urban landscape and emphasise the need for ecological knowledge to guide park design and management.

Conclusions

For the first time in north-eastern Mexico, the vegetation structure was monitored on a rural-urban gradient, where the height, cover, abundance, species richness and diversity were the parameters recorded in sites with different levels of urbanisation. It was found that all the parameters decreased with increasing urbanisation levels; thus, in accordance with the general disturbance hypothesis, the general tendency of plant distribution at the levels of urbanisation and pollution present in the MMA is to decrease. The association between environmental variables and the plant community along the urbanisation gradient was significant, the conditions of RH, DP, HI and $PM_{2.5}$ being the variables that best describe the vegetation structure in the MMA. Understanding the nature and variability of vegetation within green spaces contributes to increasing our knowledge about the distribution of the environmental services it provides and the composition of the faunal communities that depend on it. Likewise, it provides valuable information to prioritise the strategic management of the vegetation of urban green spaces so that it provides the greatest benefit for humans and biodiversity.

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Additional information

Conflict of interest

The authors have declared that no competing interests exist.

Ethical statement

No ethical statement was reported.

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Author contributions

Edmar Meléndez-Jaramillo, sampling sites selection, fieldwork, plants identification, data analysis and document writing; Laura Sánchez-Castillo, fieldwork, database compilation and document writing; Ma. Teresa de Jesús Segura Martínez, sampling sites selection, plants identification and completed document review; Uriel Jeshua Sánchez-Reyes, data analysis, results interpretation and completed document review.

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Data availability

All of the data that support the findings of this study are available in the main text.

References

- Alanís GJ (2005) El arbolado urbano en el área metropolitana de Monterrey. *Ciencia UANL* 8(1): 20–32.
- Alanís FGJ, González AD (2003) Flora Nativa Ornamental para el Área Metropolitana de Monterrey. Universidad Autónoma de Nuevo León. R. Ayuntamiento de Monterrey, México, 1–110.
- Alanís-Rodríguez E, Jiménez-Pérez J, Mora-Olivo A, Martínez-Ávalos JG, Mata-Balderas JM, Chávez-Costa AC, Rubio-Camacho EA (2015) Estructura y diversidad del matorral submontano contiguo al área metropolitana de Monterrey, Nuevo León, México. *Acta Botánica Mexicana* 113(113): 1–19. <https://doi.org/10.21829/abm113.2015.1093>
- Alberti M, Botsford E, Cohen A (2001) Quantifying the urban gradient: linking urban planning and ecology. In: Marzluff JM, Bowman R, Donnelly R (Eds) *Avian Ecology in an Urbanizing World*. Kluwer, Norwell, Massachusetts, 68–85. https://doi.org/10.1007/978-1-4615-1531-9_5
- Arreola JL, González G (1999) Análisis espectral del viento y partículas menores de 10 micrómetros (PM10) en el área Metropolitana de Monterrey México. *Revista Internacional de Contaminación Ambiental* 15(2): 95–102.
- Arroyo-Rodríguez V, Melo FP, Martínez-Ramos M, Bongers F, Chazdon RL, Meave JA, Norden N, Santos BA, Leal IR, Tabarelli M (2017) Multiple successional pathways in human-modified tropical landscapes: New insights from forest succession, forest

- fragmentation and landscape ecology research. *Biological Reviews of the Cambridge Philosophical Society* 92(1): 326–340. <https://doi.org/10.1111/brv.12231>
- Badillo CT, Garza L, Garza MCH, Zanatta MT, Caballero A (2015) Heavy Metal Content in PM2.5 Air Samples Collected in the Metropolitan Area of Monterrey, México. *Human and Ecological Risk Assessment* 21(8): 2022–2035. <https://doi.org/10.1080/10807039.2015.1017873>
- Baumgardner D, Varela S, Escobedo FJ, Chacalo A, Ochoa C (2012) The role of a peri-urban forest on air quality improvement in the Mexico City megalopolis. *Environmental Pollution* 163: 174–183. <https://doi.org/10.1016/j.envpol.2011.12.016>
- Bernhardt ES, Palmer MA (2007) Restoring streams in an urbanizing world. *Freshwater Biology* 52(4): 738–751. <https://doi.org/10.1111/j.1365-2427.2006.01718.x>
- Brower JE, Zar JH, Ende CNV (1998) *Field and Laboratory Methods for General Ecology*. McGraw-Hill Education, Boston.
- Burton ML, Samuelson LJ (2008) Influence of urbanization on riparian forest diversity and structure in the Georgia Piedmont, US. *Plant Ecology* 195(1): 99–115. <https://doi.org/10.1007/s11258-007-9305-x>
- Burton ML, Samuelson LJ, Pan S (2005) Riparian woody plant diversity and forest structure along an urban-rural gradient. *Urban Ecosystems* 8(1): 93–106. <https://doi.org/10.1007/s11252-005-1421-6>
- Cameron GN, Culley TM, Kolbe SE, Miller AI, Matter SF (2015) Effects of urbanization on herbaceous forest vegetation: The relative impacts of soil, geography, forest composition, human access, and an invasive shrub. *Urban Ecosystems* 18(4): 1051–1069. <https://doi.org/10.1007/s11252-015-0472-6>
- Carpio A, Ponce-Lopez R, Lozano-García DF (2021) Urban form, land use, and cover change and their impact on carbon emissions in the Monterrey Metropolitan area, Mexico. *Urban Climate* 39: 100947. <https://doi.org/10.1016/j.uclim.2021.100947>
- Castillo-Nava D, Elias-Santos M, López-Chuken UJ, Valdés-González A, de la Riva-Solís LG, Vargas-Pérez MP, Galán-Wong LJ, Luna-Olvera HA (2020) Heavy metals (lead, cadmium and zinc) from street dust in Monterrey, Mexico: ecological risk index. *International Journal of Environmental Science and Technology* 17: 3231–3240. <https://doi.org/10.1007/s13762-020-02649-5>
- Catford JA, Daehler CC, Murphy HT, Sheppard AW, Hardesty BD, Westcott DA, Rejmánek M, Bellingham PJ, Pergl J, Horvitz CC, Hulme PE (2012) The intermediate disturbance hypothesis and plant invasions: Implications for species richness and management. *Perspectives in Plant Ecology, Evolution and Systematics* 14(3): 231–241. <https://doi.org/10.1016/j.ppees.2011.12.002>
- Cerón JG, Cerón RM, Kahl JDW, Ramírez LE, Guarnaccia C, Aguilar CA, Montalvo RC, Anguebes F, López CU (2014) Diurnal and seasonal variation of BTEX in the air of Monterrey, Mexico: Preliminary study of sources and photochemical ozone pollution. *Air Quality, Atmosphere & Health* 8(5): 469–482. <https://doi.org/10.1007/s11869-014-0296-1>
- Chao A, Jost L (2012) Coverage-based rarefaction and extrapolation: Standardizing samples by completeness rather than size. *Ecology* 93(12): 2533–2547. <https://doi.org/10.1890/11-1952.1>
- Chao A, Jost L (2015) Estimating diversity and entropy profiles via discovery rates of new species. *Methods in Ecology and Evolution* 6(8): 873–882. <https://doi.org/10.1111/2041-210X.12349>
- Chao A, Ma KH, Hsieh TC, Chiu CH (2016) SpadeR (Species-richness prediction and diversity estimation in R): an R package in CRAN. http://chao.stat.nthu.edu.tw/wordpress/software_download/

- Christopoulou O, Polyzos S, Minetos D (2007) Peri-urban and urban forests in Greece: Obstacle or advantage to urban development? *Management of Environmental Quality* 18(4): 382–395. <https://doi.org/10.1108/14777830710753794>
- Czaja M, Kołton A, Muras P (2020) The Complex Issue of Urban Trees-Stress Factor Accumulation and Ecological Service Possibilities. *Forests* 11(9): e932. <https://doi.org/10.3390/f11090932>
- De la Maza C, Hernández J, Bown H, Rodríguez M, Escobedo F (2002) Vegetation diversity in the Santiago de Chile urban ecosystem. *Arboricultural Journal* 26(4): 347–357. <https://doi.org/10.1080/03071375.2002.9747349>
- Dolédec S, Chessel D, Gimaret-Carpentier C (2000) Niche separation in community analysis: A new method. *Ecology* 81(10): 2914–2927. [https://doi.org/10.1890/0012-9658\(2000\)081\[2914:NSICAA\]2.0.CO;2](https://doi.org/10.1890/0012-9658(2000)081[2914:NSICAA]2.0.CO;2)
- Duguay S, Eigenbrod F, Fahig L (2007) Effects of surrounding urbanization on non-native flora in small forest patches. *Landscape Ecology* 22(4): 589–599. <https://doi.org/10.1007/s10980-006-9050-x>
- Escobedo FJ, Nowak DJ, Wagner JE, De la Maza CL, Rodríguez M, Crane DE, Hernández J (2006) The socioeconomics and management of Santiago de Chile's public urban forests. *Urban Forestry & Urban Greening* 4(3–4): 105–114. <https://doi.org/10.1016/j.ufug.2005.12.002>
- Estrada-Castillón E, Villarreal-Quintanilla JÁ, Jurado-Ybarra E, Cantú-Ayala C, García-Aranda MA, Sánchez-Salas J, Jiménez-Pérez J, Pando-Moreno M (2012) Clasificación, estructura y diversidad del matorral submontano adyacente a la planicie costera del Golfo Norte en el noreste de México. *Botanical Sciences* 90(1): 37–52. <https://doi.org/10.17129/botsci.384>
- Flocks J, Escobedo F, Wade JS, Varela S, Wald C (2011) Environmental Justice Implications of Urban Tree Cover in Miami-Dade County, Florida. *Environmental Justice* 4(2): 125–134. <https://doi.org/10.1089/env.2010.0018>
- García M, Sánchez HUR, Godínez HU, Pérez A, Arias S (2012) Las inversiones térmicas y la contaminación atmosférica en la zona Metropolitana de Guadalajara (México). *Investigaciones Geográficas* (58): 9–29. <https://doi.org/10.14198/INGEO2012.58.01>
- García M, Ramírez H, Ulloa H, García O, Meulener Á, Alcalá J (2013) Concentración de contaminantes SO₂, NO₂ y correlación con H⁺, SO₄-2 y NO₃- durante la temporada de lluvias en la Zona Metropolitana de Guadalajara, Jalisco, México. *Revista Chilena de Enfermedades Respiratorias* 29(2): 81–88. <https://doi.org/10.4067/S0717-73482013000200004>
- Gavier-Pizarro GL, Radeloff VC, Stewart SI, Huebner CD, Keuler MS (2010) Housing is positively associated with invasive exotic plant species richness in New England, USA. *Ecological Applications* 20(7): 1913–1923. <https://doi.org/10.1890/09-2168.1>
- Godefroid S (2001) Temporal analysis of the Brussels flora as indicator for changing environmental quality. *Landscape and Urban Planning* 52(4): 203–224. [https://doi.org/10.1016/S0169-2046\(00\)00117-1](https://doi.org/10.1016/S0169-2046(00)00117-1)
- Godefroid S, Koedam N (2003) Identifying indicator plant species of habitat quality and invasibility as a guide for peri-urban forest management. *Biodiversity and Conservation* 12(8): 1699–1713. <https://doi.org/10.1023/A:1023606300039>
- González SO, Badillo CT, Kahl JD, Ramírez LE, Balderas RI (2011) Temporal Analysis of PM₁₀ in Metropolitan Monterrey, México. *Journal of the Air & Waste Management Association* 61(5): 573–579. <https://doi.org/10.3155/1047-3289.61.5.573>
- Grau HR, Hernández ME, Gutierrez J, Gasparri NI, Casavecchia MC, Flores-Ivaldi EE, Paolini L (2008) A peri-urban neotropical forest transition and its consequences for environmental services. *Ecology and Society* 13(1): 1–35. <https://doi.org/10.5751/ES-02434-130135>

- Grimm NB, Foster DR, Groffman PM, Grove JM, Hopkinson CS, Nadelhofer KJ, Pataki DE, Peters DPC (2008) The changing landscape: Ecosystem responses to urbanization and pollution across climate and societal gradients. *Frontiers in Ecology and the Environment* 6(5): 264–272. <https://doi.org/10.1890/070147>
- Groffman PM, Bain DJ, Band LE, Belt KT, Brush GS, Grove JM, Pouyat RV, Yesilonis IC, Zipperer WC (2003) Down by the riverside: Urban riparian ecology. *Frontiers in Ecology and the Environment* 1(6): 315–321. [https://doi.org/10.1890/1540-9295\(2003\)001\[0315:DBTRUR\]2.0.CO;2](https://doi.org/10.1890/1540-9295(2003)001[0315:DBTRUR]2.0.CO;2)
- Guariguata MR, Ostertag R (2001) Neotropical secondary forest succession: changes in structural and functional characteristics. *Forest Ecology and Management* 148(1–3):185–206. [https://doi.org/10.1016/S0378-1127\(00\)00535-1](https://doi.org/10.1016/S0378-1127(00)00535-1)
- Güler B (2020) Plant species diversity and vegetation in urban grasslands depending on disturbance levels. *Biologia* 75(9): 1231–1240. <https://doi.org/10.2478/s11756-020-00484-0>
- Hahs AK, McDonnell MJ (2006) Selecting independent measures to quantify Melbourne’s urban-rural gradient. *Landscape and Urban Planning* 78(4): 435–448. <https://doi.org/10.1016/j.landurbplan.2005.12.005>
- Hardwick SR, Toumi R, Pfeifer M, Turner EC, Nilus R, Ewers RM (2015) The relationship between leaf area index and microclimate in tropical forest and oil palm plantation: Forest disturbance drives changes in microclimate. *Agricultural and Forest Meteorology* 201: 187–195. <https://doi.org/10.1016/j.agrformet.2014.11.010>
- Hill MO (1973) Diversity and evenness: A unifying notation and its consequences. *Ecology* 54(2): 427–432. <https://doi.org/10.2307/1934352>
- Hope D, Gries C, Zhu W, Fagan WF, Redman CL, Grimm NB, Kinzig A (2003) Socioeconomics drive urban plant diversity. *Proceedings of the National Academy of Sciences of the United States of America* 100(15): 8788–8792. <https://doi.org/10.1073/pnas.1537557100>
- Hsieh TC, Ma KH, Chao A (2016) iNEXT: An R package for interpolation and extrapolation of species diversity (Hill numbers). *Methods in Ecology and Evolution* 7(12): 1451–1456. <https://doi.org/10.1111/2041-210X.12613>
- INEGI (2016) Conjunto de Datos Vectoriales de Uso de Suelo y Vegetación. Escala: 1: 250 000. Serie VI (Capa Union) Edición: 1. Instituto Nacional de Estadística y Geografía, Aguascalientes, México.
- INEGI (2021) Instituto Nacional de Estadística y Geografía. Censo de Población y Vivienda 2020. Instituto Nacional de Estadística y Geografía, México.
- IPNI (2022) International Plant Names Index. The Royal Botanic Gardens, Kew, Harvard University Herbaria & Libraries and Australian National Botanic Gardens. [Retrieved 22 June 2022] <http://www.ipni.org>
- Jim CY (2002) Heterogeneity and differentiation of the tree flora in three major land uses in Guangzhou City, China. *Annals of Forest Science* 59(1): 107–118. <https://doi.org/10.1051/forest:2001010>
- Jim CY, Chen WY (2006) Recreation-amenity use and contingent valuation of urban green space in Guangzhou, China. *Landscape and Urban Planning* 75(1–2): 81–96. <https://doi.org/10.1016/j.landurbplan.2004.08.008>
- Jim CY, Liu HT (2001) Species diversity of three major urban forest types in Guangzhou City, China. *Forest Ecology and Management* 146(1–3): 99–114. [https://doi.org/10.1016/S0378-1127\(00\)00449-7](https://doi.org/10.1016/S0378-1127(00)00449-7)
- Jiménez-Valverde A, Hortal J (2003) Las curvas de acumulación de especies y la necesidad de evaluar la calidad de los inventarios biológicos. *Revista Iberica de Aracnología* 8: 151–161.

- Johnson AL, Swan CM (2014) Drivers of Vegetation Species Diversity and Composition in Urban Ecosystems In: McCleery R, Moorman C, Peterson M (Eds) *Urban Wildlife conservation*. Springer, Boston, 75–90. https://doi.org/10.1007/978-1-4899-7500-3_5
- Kremen C (1992) Assessing the indicator properties of species assemblages for natural areas monitoring. *Ecological Applications* 2(2): 203–217. <https://doi.org/10.2307/1941776>
- Kühn I, Klotz S (2006) Urbanization and homogenization – Comparing the floras of urban and rural areas in Germany. *Biological Conservation* 127(3): 292–300. <https://doi.org/10.1016/j.biocon.2005.06.033>
- LaPaix R, Freedman B (2010) Vegetation structure and composition within urban parks of Halifax regional municipality, Nova Scotia, Canada. *Landscape and Urban Planning* 98(2): 124–135. <https://doi.org/10.1016/j.landurbplan.2010.07.019>
- Lezama JL, Graizbord B (2010) *Los Grandes Problemas de México. IV Medio Ambiente. 1a. Edición*. El Colegio de México, México, 431 pp. <https://www.jstor.org/stable/j.ctt1657t3w>
- Lienard J, Florescu I, Strigul N (2015) An Appraisal of the Classic Forest Succession Paradigm with the Shade Tolerance Index. *PLoS ONE* 10(2): e0117138. <https://doi.org/10.1371/journal.pone.0117138>
- López E, Bocco G, Mendoza M, Duhau E (2001) Predicting land-cover and land-use change in the urban fringe. *Landscape and Urban Planning* 55(4): 271–285. [https://doi.org/10.1016/S0169-2046\(01\)00160-8](https://doi.org/10.1016/S0169-2046(01)00160-8)
- Mancilla Y, Herckes P, Fraser MP, Mendoza A (2015) Secondary organic aerosol contributions to PM_{2.5} in Monterrey, Mexico: Temporal and seasonal variation. *Atmospheric Research* 153: 348–359. <https://doi.org/10.1016/j.atmosres.2014.09.009>
- ManSecak AM, Wein RW (2006) Exotic vascular plant invasiveness and forest invasibility in urban boreal forest types. *Biological Invasions* 8(8): 1651–1662. <https://doi.org/10.1007/s10530-005-5874-6>
- McKinney ML (2008) Effects of urbanization on species richness: A review of plants and animals. *Urban Ecosystems* 11(2): 161–176. <https://doi.org/10.1007/s11252-007-0045-4>
- McKinney ML (2006) Urbanization as a major cause of biotic homogenization. *Biological Conservation* 127(3): 247–260. <https://doi.org/10.1016/j.biocon.2005.09.005>
- McKinney ML (2002) Urbanization, biodiversity, and conservation. *BioScience* 52: 883–890. [https://doi.org/10.1641/0006-3568\(2002\)052\[0883:UBAC\]2.0.CO;2](https://doi.org/10.1641/0006-3568(2002)052[0883:UBAC]2.0.CO;2)
- Menchaca HL, Mercado R, Mendoza A (2015) Diurnal and seasonal variation of volatile organic compounds in the atmosphere of Monterrey, Mexico. *Atmospheric Pollution Research* 6(6): 1073–1081. <https://doi.org/10.1016/j.apr.2015.06.004>
- Moffatt SF, McLachlan SM, Kenkel NC (2004) Impacts of land use on riparian forest along an urban-rural gradient in southern Manitoba. *Plant Ecology* 174(1): 119–135. <https://doi.org/10.1023/B:VEGE.0000046055.27285.fd>
- Mohamed MOS, Neukermans G, Kairo JG, Dahdouh-Guebas F, Koedam N (2009) Mangrove forests in a peri-urban setting: the case of Mombasa (Kenya). *Wetlands Ecology and Management* 17(3): 243–255. <https://doi.org/10.1007/s11273-008-9104-8>
- Mueller-Dombois D, Ellenberg H (1974) *Aims and Methods of Vegetation Ecology*. John Wiley and Sons, New York.
- Mukherjee S, Banerjee S, Saha GK, Basu P, Aditya G (2015) Butterfly diversity in Kolkata, India: An appraisal for conservation management. *Journal of Asia-Pacific Biodiversity* 8(3): 210–221. <https://doi.org/10.1016/j.japb.2015.08.001>
- Norris CP, Hobson P, Ibsch PL (2012) Microclimate and vegetation function as indicators of forest thermodynamic efficiency. *Journal of Applied Ecology* 49(3): 562–570. <https://doi.org/10.1111/j.1365-2664.2011.02084.x>

- Oksanen J, Blanchet FG, Friendly M, Kindt R, Legendre P, McGlenn D, Minchin PR, O'Hara RB, Simpson GL, Solymos P, Stevens MHH, Szoecs E, Wagner H (2019) vegan: Community Ecology Package. R package version 2.5–4. <https://CRAN.R-project.org/package=vegan>
- Orta ST, Ochoa AC, Carrizalez L, Varela JA, Pérez FJ, Pruneda LG, Torres A, Guzmán JL, Pérez IN (2016) Persistent organic pollutants and heavy metal concentrations in soil from the Metropolitan Area of Monterrey, Nuevo Leon, Mexico. *Archives of Environmental Contamination and Toxicology* 70(3): 452–463. <https://doi.org/10.1007/s00244-015-0239-3>
- Parr CL (2012) Unpacking the impoverished nature of secondary forests. *Journal of Animal Ecology* 81(5): 937–939. <https://doi.org/10.1111/j.1365-2656.2012.02016.x>
- Paul MJ, Meyer JL (2001) Streams in the Urban Landscape. *Annual Review of Ecology and Systematics* 32(1): 333–365. <https://doi.org/10.1146/annurev.ecolsys.32.081501.114040>
- Pedlowski MA, Da Silva VAC, Adell JJC, Heynen NC (2002) Urban forest and environmental inequality in Campos dos Goytacazes, Rio de Janeiro, Brazil. *Urban Ecosystems* 6(1/2): 9–20. <https://doi.org/10.1023/A:1025910528583>
- Pennington DN, Hansel JR, Gorchov DL (2010) Urbanization and riparian forest woody communities: Diversity, composition, and structure within a metropolitan landscape. *Biological Conservation* 143(1): 182–194. <https://doi.org/10.1016/j.biocon.2009.10.002>
- Porter EE, Forschner BR, Blair RB (2001) Woody vegetation and canopy fragmentation along a forest-to-urban gradient. *Urban Ecosystems* 5(2): 131–151. <https://doi.org/10.1023/A:1022391721622>
- Puric-Mladenovic D, Kenney WA, Csillag F (2000) Land development pressure on peri-urban forests: A case study in the Regional Municipality of York. *Forestry Chronicle* 76(2): 247–250. <https://doi.org/10.5558/tfc76247-2>
- Quantum GIS Development Team (2018) Quantum GIS Geographic Information System. Open Source Geospatial Foundation Project. <https://qgis.org/es/site/>
- R Development Core Team (2019) R: a language and environment for statistical computing. R Foundation for Statistical Computing, Vienna.
- Ramírez A (2006) *Ecología: Métodos de Muestreo y Análisis de Poblaciones y Comunidades*. Editorial Pontificia Universidad Javeriana, Bogotá.
- Renaud V, Innes JL, Dobbertin M, Rebetez M (2010) Comparison between open-site and below-canopy climatic conditions in Switzerland for different types of forests over 10 years (1998–2007). *Theoretical and Applied Climatology* 105(1–2): 119–127. <https://doi.org/10.1007/s00704-010-0361-0>
- Richardson DM, Holmesl PM, Esler KJ, Galatowitsch SM, Stromberg JC, Kirkman SP, Pysek P, Hobbs RJ (2007) Riparian vegetation: Degradation, alien plant invasions, and restoration prospects. *Diversity & Distributions* 13(1): 126–139. <https://doi.org/10.1111/j.1366-9516.2006.00314.x>
- Sanchez-Castillo L, Kubota T, Cantú I, Moriyama T (2016) A probability method of rainfall warning for sediment-related disaster in developing countries: A case study in Sierra Madre Oriental, Mexico. *Natural Hazards* 85(3): 1893–1906. <https://doi.org/10.1007/s11069-016-2669-2>
- Schwartz MW, Thorne JH, Viers JH (2006) Biotic homogenization of the California flora in urban and urbanizing regions. *Biological Conservation* 127(3): 282–291. <https://doi.org/10.1016/j.biocon.2005.05.017>
- Sisto NP, Ramírez AI, Aguilar I, Magaña V (2016) Climate threats, water supply vulnerability and the risk of a water crisis in the Monterrey Metropolitan Area (Northeastern

- Mexico). *Physics and Chemistry of the Earth Parts A/B/C* 91: 2–9. <https://doi.org/10.1016/j.pce.2015.08.015>
- Stewart GH, Meurk CD, Ignatieva ME, Buckley HL, Magueur A, Case BS, Parker M (2009) *URban Biotopes of Aotearoa New Zealand (URBANZ) II: Floristics, biodiversity and conservation values of urban residential and public woodlands*, Christchurch. *Urban Forestry & Urban Greening* 8(3): 149–162. <https://doi.org/10.1016/j.ufug.2009.06.004>
- Stubbendieck JL, Hatch SL, Landholt LM (2003) *North American Wildland Plants: A Field Guide* (6th ed.). University of Nebraska Press, Nebraska.
- Styers DM, Chappelka AH, Marzen LJ, Somers GL (2010) Scale matters: Indicators of ecological health along the urban-rural interface near Columbus, Georgia. *Ecological Indicators* 10(2): 224–233. <https://doi.org/10.1016/j.ecolind.2009.04.018>
- Tang J, Wang L, Yao Z (2008) Analyses of urban landscape dynamics using multi-temporal satellite images: A comparison of two petroleum-oriented cities. *Landscape and Urban Planning* 87(4): 269–278. <https://doi.org/10.1016/j.landurbplan.2008.06.011>
- TIBCO Software Inc (2017) *Statistica (data analysis software system): version 13.3*. Palo Alto, CA. <https://www.tibco.com/products/tibco-statistica>
- Turner WR, Nakamura T, Dinetti M (2004) Global urbanization and the separation of humans from nature. *BioScience* 54: 585–590. [https://doi.org/10.1641/0006-3568\(2004\)054\[0585:GUATSO\]2.0.CO;2](https://doi.org/10.1641/0006-3568(2004)054[0585:GUATSO]2.0.CO;2)
- Turner K, Lefler L, Freedman B (2005) Plant communities of selected urbanized areas of Halifax, Nova Scotia, Canada. *Landscape and Urban Planning* 71(2–4): 191–206. <https://doi.org/10.1016/j.landurbplan.2004.03.003>
- Vilà M, Esinar JL, Hejda M, Hulme PE, Jarošík V, Maron JL, Pergl J, Schaffner U, Sun Y, Pyšek P (2011) Ecological impacts of invasive alien plants: A meta-analysis of their effects on species, communities, and ecosystems. *Ecology Letters* 14(7): 702–708. <https://doi.org/10.1111/j.1461-0248.2011.01628.x>
- Villaseñor JL, Espinosa-García FJ (2004) The alien flowering plants of Mexico. *Diversity & Distributions* 10(2): 113–123. <https://doi.org/10.1111/j.1366-9516.2004.00059.x>
- Vlahov D, Galea S (2002) Urbanization, urbanicity, and health. *Journal of Urban Health* 79(90001, Suppl. 1): 1–12. https://doi.org/10.1093/jurban/79.suppl_1.S1
- Ybáñez E, Barboza C (2017) Trayectorias recientes de la migración interna en la Zona Metropolitana de Monterrey: Características, orígenes y destinos a nivel municipal, 2010. *Estudios Demográficos y Urbanos* 32(2): 245–281. <https://doi.org/10.24201/edu.v32i2.1722>
- Zegeye G, Erifo S, Addis G, Gebre GG (2023) Economic valuation of urban forest using contingent valuation method: The case of Hawassa city, Ethiopia. *Trees, Forests and People* 12: e100398. <https://doi.org/10.1016/j.tfp.2023.100398>
- Zhao M, Escobedo FJ, Staudhammer C (2010) Spatial patterns of a subtropical, coastal urban forest: Implications for land tenure, hurricanes, and invasives. *Urban Forestry & Urban Greening* 9(3): 205–214. <https://doi.org/10.1016/j.ufug.2010.01.008>
- Zurita ZO, Elizondo ER (2009) *Guía de árboles y otras plantas nativas en la zona metropolitana de Monterrey*. Fondo Editorial de Nuevo León, Monterrey N.L., México.