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Relationship between tree cover and tree diversity in urban parks in a semiarid city in Mexico

Lorena Marion Martínez Mompha[‡], Carlos Renato Ramos Palacios[§], José Arturo de Nova Vázquez[|], Fredy Alvarado[¶], Leonardo Ernesto Márquez Mireles[#], Roberto Briones Gallardo[¤]

‡ Programa Multidisciplinario de Posgrado en Ciencias Ambientales, Universidad Autónoma de San Luis Potosí, San Luis Potosí, Mexico
§ Facultad del Hábitat, Universidad Autónoma de San Luis Potosí, San Luis Potosí, Mexico
| Instituto de Investigación en Zonas Desérticas, Facultad de Agronomía, Universidad Autónoma de San Luis Potosí, San Luis Potosí, Mexico
¶ Instituto Potosino de Investigación Científica y Tecnológica, División de Ciencias Ambientales., San Luis Potosí, Mexico
# Facultad de Ciencias Sociales y Humanidades. Universidad Autónoma de San Luis Potosí, San Luis Potosí, Mexico
¤ Instituto de Metalurgía, Facultad de Ingeniería, San Luis Potosí, Universidad Autónoma de San Luis Potosí, San Luis Potosí, Mexico

Corresponding author: Lorena Marion Martínez Mompha (a201629@alumnos.uaslp.mx)

Abstract

Trees play a crucial role in urban parks and contribute significantly to ecosystem services. However, their composition is often dominated by introduced species, which can reduce potential environmental benefits. This study assessed tree cover density, species diversity, and indirect carbon sequestration in three urban parks in San Luis Potosí, serving as a case study for a semi-arid city. The data was obtained from randomly selected sampling sites. Morphometric characteristics and carbon storage were analyzed using the i-Tree program, while species diversity was evaluated through the Shannon and Simpson indices using RStudio™. Results showed that tree cover in all parks was below 50%, with fewer than four species making up most of the population. Of the total species identified, 60% were non-native and 40% native. The average values for the Shannon and Simpson indices were 2.2 and 0.8, respectively. This information can support informed decision-making for reforestation strategies aimed at enhancing tree density, increasing biodiversity, and maximizing the ecological functions of urban parks.

Keywords

SIG, urban parks, i-Tree software, carbon sequestration

Introduction

In an urban park (UP), environmental and ecological quality lie fundamentally in the attributes of the structure and diversity of its vegetation cover. One of the most outstanding plant forms of cover in a park situation is arboreal. Most internationally reported studies show that tree cover is one of the principles for the ecological study of urban parks (Li et al. 2005, Nix et al. 2022, Ortega Rosas et al. 2022, Wang et al. 2023). In recent literature, the density and composition of tree elements are key ecological studies for determining carbon sequestration in UPs (Shehzad et al. 2023, Reyes Chan et al. 2024), oxygen production (Chapin et al. 2011), the mitigation of the heat island phenomenon (Wang et al. 2023, Reyes Chan et al. 2024), the retention of atmospheric pollutants (Heshani and Winijkul 2022, Zhou et al. 2024), among others. Parks also favor the creation of habitats for diverse biological species, since a positive relationship has been demonstrated between plant diversity and different animal species (Correa et al. 2018, Yang et al. 2020, Han et al. 2021, Kaushik et al. 2021), and the same has been found in fragments with green covers (Dunn and Heneghan 2011, Matthies et al. 2017). Thus, the environmental functions of urban parks depend largely on tree diversity (Farrell et al. 2022), and as this increases, it can also contribute to dealing with possible pests or effects of environmental stress (Alvey 2006), such as increased temperature, pollution of the atmosphere, lack of water and little space (Aronson et al. 2017).

However, in the literature, it has been found that the tree composition of UPs is represented by monocultures or they tend to be dominated by only a few species, generally introduced (Figueroa et al. 2018, Kumar Pandey and Kumar 2018, Olokeogun et al. 2020, Valle 2018). That is why the selection of species depends entirely on the people in charge of these areas, who tend to give greater weight to aesthetics rather than to the ecosystem functioning of UPs. It has been observed that the vision of the benefits that trees can bring is very limited, for example, in the city of Fortaleza, Brazil, the selection of species is due to their ability to provide shade or fruit (Moro et al. 2014). This pattern may result in diminished ecosystem services in these areas and is evident in both temperate cities (Domínguez Liévano 2023, Ortega Rosas et al. 2022), and arid and semiarid zones (Bano et al. 2023, MOSYAFTIANI et al. 2022). Although at any latitude the green cover of urban parks is planted, in semiarid cities, where native vegetation is not always high in tree individuals, the use of introduced species allows the increase of tree cover and its richness (Augustinus et al. 2024,Nielsen et al. 2013, Zipperer et al. 1997). However, this increase can generate problems since introduced species can become biological invaders, displacing and reducing the growth of native species (Morgenroth et al. 2016); they can produce important environmental or ecological changes and, therefore, promote the decline of biodiversity at the regional level (McKinney 2006). On the other hand, the difficulty in choosing the best species to obtain various desired benefits is understandable, since a balance must be found between the ecological management of these areas and the intended use of the place. Another obstacle is that specialized literature on ecological issues in urban parks has more reports in temperate cities and high latitudes than in arid or semiarid regions (El-Metwally et al. 2025).

In the case of Mexico, the country is making slow progress on these research topics where research is predominantly published in national journals, unlike other Latin American countries such as Brazil, Colombia, or Chile (Flores et al. 2022). This is the case of the city of San Luis Potosí, where there are some studies on green areas in the city where the survival of native versus introduced species has been evaluated in Tangamanga I Park, where native species had a higher survival rate (Gómez Briones 2005); official information on parks and gardens has been collected until 2010 (Rodríguez-Rangel 2010) and more currently, the effective green cover of green areas within the city has been evaluated (Ramos-Palacios et al. 2024). However, none of them have evaluated ecological factors such as tree diversity in urban parks. In this way, the present work studied the outstanding parks of the city of San Luis Potosí, Mexico, which are semiarid and where the creation of UPs has not been parallel to the unregulated urban growth it has experienced in recent years.

Materials and methods

General characteristics of study sites

The three UPs studied are in the city of San Luis Potosí (Fig. 1), which has an area of 1,787.7 km2 and a population of 845,941 inhabitants (Instituto Nacional de Estadística, Geografía e Informática (Inegi). 2020). The type of climate is dry temperate with rains in summer (BS0kW) according to the Köppen classification modified by Enriqueta García (García 1998), with an average annual temperature of 17.5°C. The average annual potential evaporation in the city is 2038.7 mm and an average annual rainfall of 402.6 mm (Comisión Nacional del Agua. [Conagua]. 2020).

The city is bounded by the Sierra de San Miguelito to the west and southeast and the Sierra de Álvarez to the east (Contreras Servín and Galindo Mendoza 2008), and the surrounding vegetation is composed of secondary vegetation of natural grassland, crassicual scrub and pine, as well as irrigated and stormy agriculture (Galindo Mendoza 2018).

UPs were selected for their size and capacity to concentrate tree cover. These parks, called Tangamanga I and II, are 411 and 189 ha, respectively, both created in the 1980s (Centros Estatales de Cultura y Recreación Tangamanga I y II 2025) and managed by the state government. The third, with a smaller area, is the Juan H. Sánchez Park, popularly called Morales, opened in 1924 with 16 hectares (Ruiz 2022), managed by the municipality.

Satellite image analysis and tree cover sampling

Initially, a satellite image of the city was obtained using the Planet platform (Planet Labs PBC, CC BY-NC-SA 2.0, https://www.planet.com/) dated May 5, 2024. Based on this photograph, with 3 m resolution, with ortho-rectification, reflectance correction on a plane and free of cloud cover, the land use surfaces within each UPs were classified. This date was chosen to determine the cover of the trees with respect to other green covers and land uses (Fig. 2).

For determining tree cover, the surface of each park in the satellite image was first delimited with the Inegi layer of urban localities 2023, thus obtaining the areas of the parks: 198.9 ha for Tangamanga I, 187.1 ha for Tangamanga II and 17.6 ha for Morales. This delimitation was then used to process the image in the Rstudio™ program version 4.3.2, where the NDVI value was calculated for each park, obtaining the types of land use of trees, grass, bare ground, body of water and construction. Using the ArcMap 10.4.1 program, the surface area of each land use was calculated, and a network of circles of 1,000 m2 each was drawn on the plan of each park. This last calculation allowed the definition of four different categories of tree cover whose ranges were 0 to 25% (very low density), 25.1 to 50% (low density), 50.1 to 75% (high density), 75.1 to 100% (very high density).

Then, using a code in the Arcmap Phyton viewer, the sampling sites (fixed circles) were determined using a random system of the data. In this way, five random sampling points were selected for each category in Morales with a distance between circles of the same category of 50 m, and six points in Tangamanga I and II with 200 m. Thus, 68 sampling sites were obtained for the three parks, and the coordinates of their centroids were acquired through the Google Earth program.

To define each UP sampling site, a point was assigned inside the tree, and the centroid point was georeferenced with a Garmi GPS. From the centroid, the distance of 17.8 m to the four cardinal points was measured to delimit the circle of the sampling site and to measure 1,000 m2. The measure that defines the count of each specimen within the site is that each one will have a minimum height of 1.3 m, that is, measured from the trunk to the height of the chest. Thus, 489 individuals were measured for Tangamanga I, 317 for Tangamanga II and 446 for Morales. The total height of each tree per sampling site was measured using a clinometer (Forestry suppliers), the height from the ground to the crown with a state (Geosurv), the percentage of lost crown and dead branches through direct observation and the area where the crown was projected. For the latter, the orthogonal projection of the cup towards the ground was used, calculating the degree of shadow cast in the form of an ellipse (Benavides Meza and Fernández Grandizo 2012). Once the measurements were gathered, a general database was developed and fed into the i-Tree Eco v.6. program, which estimated the sequestration of raw carbon for each category of tree cover and each park.

Statistical analysis

The measurement data was adapted and imported into the RStudio™ program version 4.3.2 to determine richness, abundance, and the Shannon and Simpson indices. Then, linear models were calculated to evaluate the relationship between these variables and tree cover density per UP. Diversity and richness indices were also compared between the different categories of coverage within the same park, as well as between them. For this purpose, ANOVAs and a Tukey test were performed to determine the differences between these. For the case of non-normal data, the Kruskal-Wallis test was used and Dunn's test with Bonferroni correction was employed as a post hoc test. Linear models without mixed effects were utilized to determine if the percentage of tree cover in each park was influenced by the number of species or individuals. An Non-metric Multidimensional Scaling (NMDS) analysis was also carried out on tree composition by densities and by parks.

In addition, a distance-weighted inverse interpolation (IIDW) model was developed using the RStudio program. This model requires the greatest similarity between neighboring points and vice versa with distance.

Results

According to the analysis of the satellite image, Tangamanga I park showed a tree cover of 22.3%, Tangamanga II 37.04%, and Morales 40.7% across their respective areas. Regarding land uses other than tree cover, in Tangamanga I the predominant type was bare soil, accounting for 56.3% of the surface, followed by built-up areas with 12.28%. On the other hand, in Tangamanga II, the largest proportion corresponded to grass, with 29.6%, while built-up areas represented 25%. In Morales Park, grass dominated the coverage with 32.3%, followed by built-up areas covering 27%.

With the sampling of the trees, 32 species were identified between the three parks (Table 1), where the accumulation curve indicates that the plateau of the curve occurred when this figure in richness was reached. The cumulative value for Tangmanga I was 23 species, for Tangamanga II 15 species and Morales 22 species (Fig. 3). The accumulated value for Tangmanga I was 23 species, for Tangamanga II 15 species and Morales 22 species (Fig. 3). The most abundant species were Eucalyptus camaldulensis, which represented almost 30% of the total number of individuals, followed by Casuarina equisetifolia with 18.4%, and Fraxinus uhdei with 13.9%.

Together, these three species accounted for 58.9% of registered individuals. For Tangamanga I, Eucalyptus camaldulensis represented 28.8%, Casuarina equisetifolia 20.9% and Fraxinus uhdei 10.6%, totaling 60.3%. In Tangamanga II, Eucalyptus camaldulensis stood out with 25.9%, Schinus molle with 19.2%, Casuarina equisetifolia with 15.1% and Neltuma laevigata 12.9%, which together represented 73.2% of the individuals. In this last park, there is an area that concentrates numerous mosques.

Finally, in Morales Park, eucalyptus represented 27.4% of the total, followed by Fraxinus uhdei with 26.2% and Casuarina equisetifolia with 17.9%, accumulating 71.5% among the three.

The origin of the arboreal individuals of the three UPs are mostly introduced species, covering 62.0-65.8% of the individuals, leaving the remaining 38-34.2% to native species. In Tangamanga I, the very high density accounted for the highest number of introduced species (66.6%), followed by the very low density (61.5%). In Tangamanga II, the density with the highest number of introduced species was low (66.6%), followed by very low (60%). In the case of Morales, the category with high density had 80% of introduced species, followed by low and very low densities (71.4% and 73.3% respectively).

In the statistical analysis, the current richness between density categories and between parks showed no significant differences, except between the low densities of Tangamanga II and Morales, which are different according to the Tukey test (diff = -3.1, p = 0.014), with the lower densities of Morales Park having greater richness.

Regarding the results of the linear models, the only significant relationships were for Tangamanga I (p-value = 0.008) and Tangamanga II (p-value= 0.04), in which the percentage of tree cover increased with respect to the number of individuals. In addition, Tangamanga II also presented a significant relationship (p-value = 0.05), between the increase in the number of species and the percentage of coverage (Fig. 4).

The NMDS obtained a stress of 0.126 and it was observed that most of the sites sampled for Tangamanga I and II are similar, regardless of the density category. On the other hand, the Morales sites are more distanced, and the upper and higher categories are grouped a bit, while the low and very low categories are more dispersed. (Fig. 5)

In terms of tree diversity, the Shannon index reflected a moderately diverse community (2.34), while the Simpson index indicates low diversity (0.86). The results for each park do not differ from each other, Shannon's index was 2.1 for Morales and Tangamanga II, and 2.2 for Tangamanga I; Simpson's index was 0.8 for the three parks. This may explain the lack of significant differences between parks, however, if differences were found when analyzing the same categories between parks. This is the case of the low density of Tangamanga II and Morales, where the Shannon index was higher for Morales (diff = 0.81, p-value = 0.014), as well as for the Simpson index (Z = 2.65, p-value = 0.024), implying that the low density of Morales had greater diversity than that of Tangamanga II. Regarding the interpolation of these indices, although the spatialized result was obtained, the validation of the model indicates that it does not fit the data well.

The park with the lowest amount of sequestered carbon was Tangamanga II with 1323.72 kg year-1 ha-1 compared to Tangamanga I, which sequestered 2995.25 kg year-1 ha-1 and Morales with 3358.91 kg year-1 ha-1. The statistical analysis showed that there was a significant difference between Tangamanga II park with Morales (p-value 0.001) and with Tangamanga I (p-value = 0.0048).

As for the categories for all the parks, the sequestration of the lowest category was lower for the upper category (p-value = 0.0012) and very high (p-value = 0.0048); likewise, the lower category had a lower carbon sequestration than the high and very high category (p value= 0.005 and p-value = 0.0169, respectively). Differences were also observed between the higher density of Morales and Tangamanga I parks compared to Tangamanga II, where the density of the latter indicated a lower amount of sequestered carbon (p-value = 0, p-value .0001, respectively). Particularly for the density categories within each park, only Morales presented significant differences in the amount of carbon sequestered between the different density categories where it was observed that the lowest density has a lower sequestration compared to the highest (p-value = 0.0115) and the low density compared to the high and very high density (0.0005, p-value = 0).

Discussion

Urban parks represent the most extensive green infrastructure compared to other types of green space in the city of San Luis Potosí (Ramos-Palacios et al. 2024). However, based on the results of the present investigation, tree cover was very low, as values below 50% were obtained on the surfaces of each park. In addition, it was documented that Morales Park, although it is the smallest, was the one with the highest values and in Tangamanga I the lowest. This result could be due to the continuous management of the trees presented by the two large parks in the city, as well as the relevance that other types of areas have within them, such as bare ground, grass and built areas, unlike Morales Park. This can have important effects on the potential environmental services of each park, since healthy trees with an extensive canopy have been found to provide greater benefits (Hintural et al. 2024).

Regarding the richness found in the parks, it was very low, while with the metropolitan area of San Luis Potosí, 133 different species were found in street trees, with 69% represented by 10 species (Ramos-Palaciós 2019). In the case of the parks studied, not only was the number of species for all 3 UPs much lower with 32 species, but only three species accounted for more than 50% of the individuals. This type of pattern can be observed in many of the country's green areas, for example, in the “San Juan de Aragón Forest” in Mexico City, where a richness of 12 species was found on an area of 114 ha and only 5 species represent 76.25% of the total (Saavedra-Romero et al. 2016). Also in Linares, Nuevo León, where only 21 species were found in 14 sampled parks (Leal Elizondo et al. 2018). However, there are also examples of places where with little surface area, higher levels of richness and a better balance in species abundance have been achieved, such as the Lurie Garden in Chicago, which has an ecological management focused on promoting biodiversity, so that in 2 ha you can find 222 species of plants, of which 26 are woody (Aronson et al. 2017).

On the other hand, the presence of introduced species in the three parks studied stood out. More than 50% were non-native species, with Morales having the highest proportion of these. The trees that stand out were Eucalyptus camaldulensis and Casuarina equisetifolia, two species native to Oceania that were favored for their rapid growth (

Yáñez-Espinosa et al. 2019). However, it has been demonstrated that both species can create problems because they are susceptible to pests, damage urban infrastructure with their roots, high water consumption, allergenic effects and low efficiency in retaining particulate matter (Reyes-Agüero and Ortiz Almendariz 2022).

Another interesting fact was that despite the difference in areas and the percentage of tree cover between the parks, there were no significant differences in the value of richness. The only inequality found was a greater richness present in the low density of Morales compared to that of Tangamanga II. This can be explained by the positive relationships found for Tangamanga II, where both the number of individuals and species increased along with the percentage of coverage, so that these two variables will be lower at low densities. In the case of Morales, this relationship was not found, so a greater number of species can be found regardless of the category, that is, the percentage of coverage. Nor was a positive relationship found between tree cover and diversity, measured using the Shannon and Simpson indices. Also, in the NMDS, clustering was observed between the Tangamanga I and II sites, regardless of density categories, while the Morales sites were scattered, especially those with low density. These results can be explained by the different management of Tangamanga I and II, who depend on the state government, while Morales depends on the municipal government.

The low richness and dominance of certain species was reflected in the diversity indices obtained, where the Shannon index evaluated the diversity of the community as moderate, which could be due to the equity observed in non-dominant species (Carmona-Galindo and Carmona 2013). The Simpson index took into account the weighted average, so it gave weight to each species according to its proportion, which explains the high value of the index, since there are three species that had the highest proportion within the community, indicating that diversity is low. Similar values can be found in other regions of the world, such as the city of Herat, Afghanistan, where two parks were studied and presented the same patterns as the study sites, with two or three species dominating at least 50% of the individuals, and a richness of less than 20 (Rahmani et al. 2024). Another case was in Portland, Oregon, where a difference was established between parks for recreational use, which have sports and play areas, and those that encourage walking and rest and have fewer buildings. The latter showed higher diversity values than the former (Talal and Santelmann 2019).

This result could be transferred to the parks studied, which have a greater recreational use, and this is one of the reasons why they have such low diversity, which leads us to reflect on the balance to be found between the uses given to parks and the ecological management they need to increase diversity and thus their environmental benefits.

All this data set led us to observe the quality of ecosystem services, including carbon sequestration, which was 10238.5 kg ha-1 of carbon per year (1,024 kg m-2 y-1) for all the parks. This amount was lower than that found in China, where the average of the areas studied was 2.06 KgC m-2 y-1 (Wang et al. 2021). As for the sequestration of each area, Tangamanga II Park sequestered a smaller amount of carbon, probably due to the reduced number of individuals found in this space compared to the other two parks. The importance in terms of the number of trees to increase this service was established in other studies where the amount of carbon sequestration occurred in areas with the highest density of trees (Kim et al. 2024, De la Sota et al. 2019), other elements to take into account is that evergreen trees have a greater sequestration capacity than deciduous trees, as well as the size of the individuals that the larger the greater the greater the efficiency (Kim et al. 2024,Wang et al. 2021). On the other hand, Wang et al. 2021, showed that the efficiency of carbon sequestration in parks with a higher density of trees did not differ from parks with an average density. This may explain why in the parks studied there were only differences between the lowest and highest categories, leaving aside the two middle categories.

Urban parks around the world present this type of pattern, even if certain differences can be found depending on the type of climate in each region. The average tree cover in tropical climates reached 55%, which can be explained by the fact that the same climatic conditions allow and facilitate the growth of plant cover in these areas. However, coverage in tropical and temperate cities tends to be lower than natural vegetation because in these regions the lost of green cover is higher due to the extensive urbanisation. In contrast, although it seems that the average of urban parks reached 50% of coverage in dry climates, this was greater than in natural areas since more plant cover is planted. Fig. 6 shows the values of tree cover, richness, and the origin of species (native or introduced), obtained from publications in semiarid cities. In the case of the parks studied, the fact that the coverage does not reach the average of dry areas can be explained by the lack of importance given to it in political agendas, where it is preferred to continue building recreational areas than to create a management plan for these places.

In the same way, the average richness found in parks in different regions is between 20 and 30 species, which can be considered low in many places, especially in tropical regions, where the tree diversity of natural areas is much higher (Silva et al. 2020). On the other hand, for dry regions such as those in study areas, high richness is often due to the incorporation of elements from wetter areas, which may have invasive potential, putting native diversity at risk.

In the case of the parks studied, Morales, which has a smaller extension, had a greater number of species than Tangamanga I and II, which have an extension up to 10 times larger, but the same proportion is not seen in terms of richness. It is important to emphasize that the surface area of these green areas is not the main element to consider when calculating tree diversity or cover. There are several examples where many trees were observed in relatively small areas (Aronson et al. 2017), so it is possible to aim to increase tree diversity if a selection method is created to consider the social and environmental needs of places.

It is well known that the use of exotic species in urban areas has been increasing (Brundu et al. 2020), however, it attracts attention as in tropical climates where plant diversity is very high, so it seems unnecessary (Silva et al. 2020). In fact, it has been found that tropical areas can have a richness and abundance of exotic species that correspond to more than 80% (idem). Overall, this pattern has a historical context, where colonization played an important role in selecting ornamental and rapidly growing species in certain places, as well as achieving a certain aesthetic (Reyes-Agüero and Ortiz Almendariz 2022). Regarding diversity indices, these allow us to consider both the number of individuals of each species and their abundance. It may seem strange that the dry climate will have a higher average index, however, this may be due to the need in these places to introduce attractive species from humid places, unlike other climates.

With these results, it can be suggested the planning of parks is still based on aesthetic rather than ecological standards (Barrantes-Sotela 2019), when their design or renovation should also consider an ecological vision (Romero and Toledo 2000). Not only this, but it must take into account the context of the parks and the objectives sought in these places, both environmentally and socially, to achieve good management of the place (Aly and Dimitrijevic 2022). Some changes that could improve these areas are to be considered following the Santamour rule, which proposes that tree diversity should be governed so that the total number of individuals does not correspond to more than 10% of a species, 20% of a genus or 30% of a family (Santamour 1990).

Also give preference to native species and if introduced species are used, it is necessary to ensure that there is no risk of invasion (Brundu et al. 2020). With regard to carbon sequestration, perennial trees allow carbon to be captured throughout the year, including the winter season, and those with average size do so as they grow (Wang et al. 2021), these two characteristics being important when selecting tree individuals. Another important point that must be taken into account is the role of the planted landscape that generates conditions for the spontaneous growth of plants (Chang et al. 2021), which adapt easily to the area, have low maintenance costs (Ilie and Cosmulescu 2023) and become an option for soil restoration and revegetation (Ilunga wa Ilunga 2015). Although urban parks are constantly being intervened, we must ensure that this does not harm the possibility of spontaneous plant growth, and the very cycle of these ecosystems. Finally, it is essential to ensure the environmental benefits of urban parks that benefit users and the city so much, so we must also ensure tree cover that provides these functions. In addition to increasing studies in arid and semiarid cities within these research lines.

Conclusions

The function of the UPs studied is social and public, and environmental functions have been neglected, causing a decrease in tree cover and therefore compromising ecosystem services. In the same way, tree diversity needs to be increased, avoiding that few species monopolize richness, giving priority to native species or in any case if they are introduced that, in addition to being adapted to the region, do not generate ecological alterations to the environment. Although the parks studied do not have significant tree diversity, there are other parks in semiarid areas where diversity may be higher due to the management and goals set by this type of green areas. It is also important to mention that this is a first starting point to give greater visibility to the problems observed in the parks studied where the rehabilitation of green areas is necessary, recognizing that these spaces provide ecosystem functions and services for the urban system. In addition, it highlights the lack of studies and information in arid and semiarid urban areas, places with a lack of water resources and where there is less tree cover than in other latitudes, issues that must be considered during the planning and management of urban parks.

Conflicts of interest

The authors have declared that no competing interests exist.

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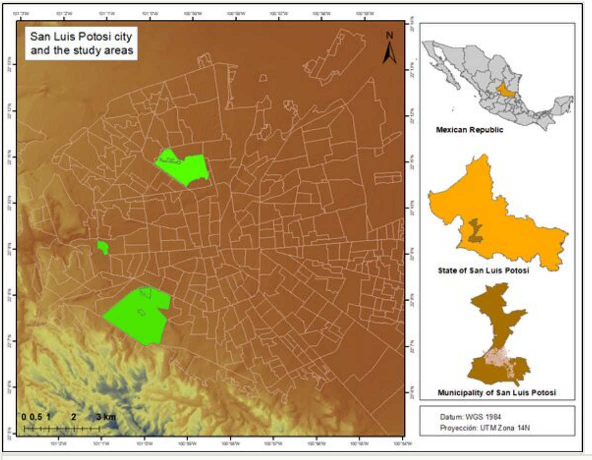

Figure 1.

Location of the urban parks of San Luis Potosi city, Mexico.

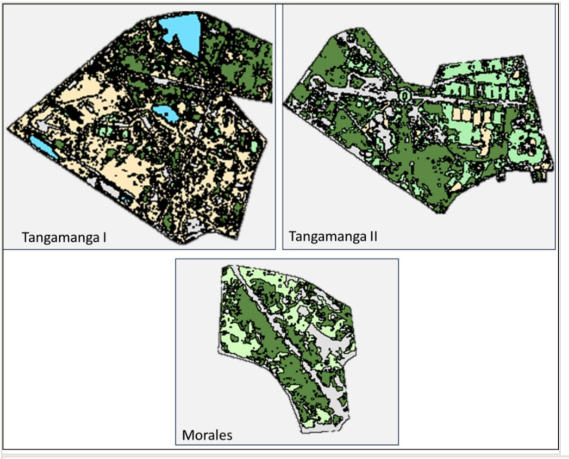

Figure 2.

Land use clasification of the three urban parks studied.

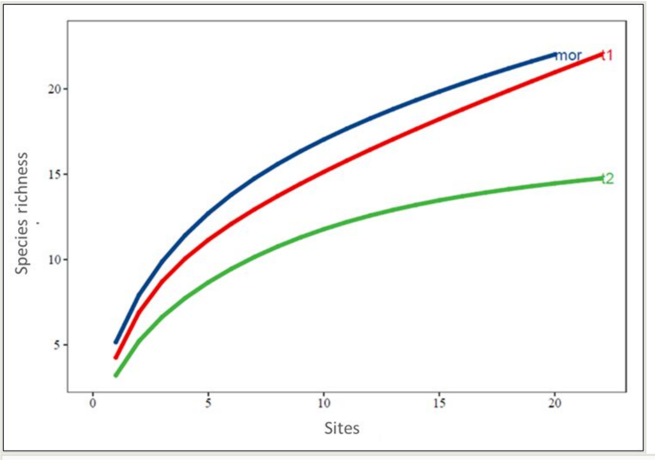

Figure 3.

Species accumulation curve for each park studied. The Mor abreviation is the Morales park, ti means Tangamanga I park and t2 the Tangamanga II park.

https://doi.org/10.3897/arphapreprints.e167706

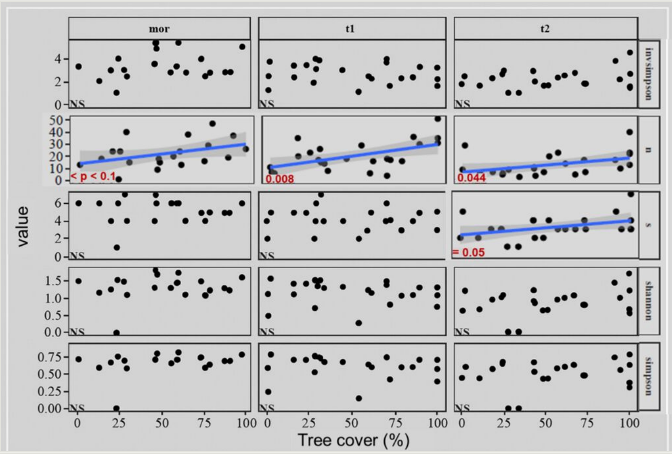

Figure 4.

Linear models between inverse Simspson, number of trees, trees species, Shannon index, Simpson index and tree cover density.

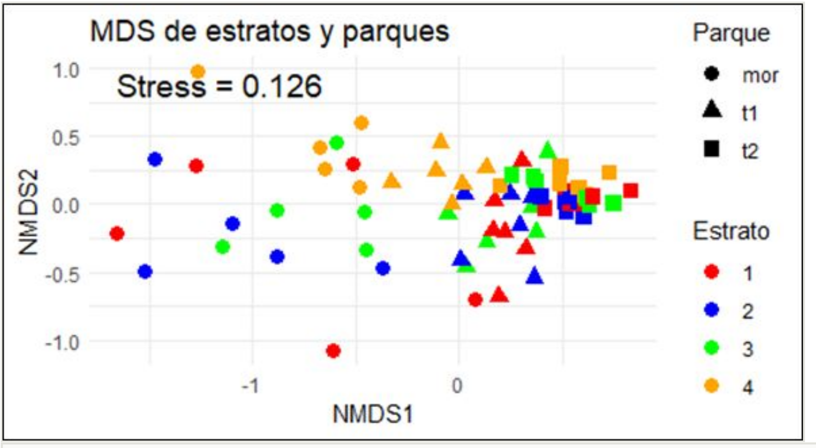

Figure 5.

The Non-metric Multidimensional Scaling (NMDS) analysis of species composition for the diffrents categories and parks.

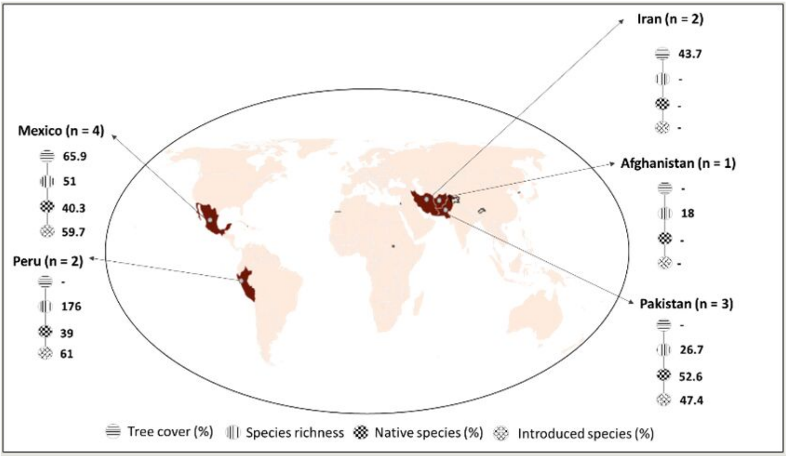

Figure 6.

Tree cover, species richness, native and introduced species of different semi arid countries.

Table 1.

Scientific name and common name of the species trees identified in each park.

No.	Scientific name	Common name	Tangamanga I	Tangamanga II	Morales
1	*Acer negundo* L.	Boxelder	X		X
2	*Buddleja cordata* Kunth	Tepozan	X		
3	*Carya illinoinensis* (Wangenh.) K.Koch	Juglan	X	X	
4	*Casuarina equisetifolia*	Beef wood	X	X	X
5	*Cupressus macrocarpa* Hartw.	Cedar	X	X	X
6	*Cupressus sempervirens* L.	Italian cypress	X		
7	*Dodoneae Viscosa L.* Jacq.	Chapulixtle	X		
8	*Eucalyptus camaldulensis* Dehnh.	River red gum	X	X	X
9	*Eucalyptus cinerea* F.Muell. ex Benth.	Silver dollar tree	X		
10	*Ficus benjamina*	Malayan banyan			X
11	*Fraxinus pennsylvanica* Marshall	Red ash			X
12	*Fraxinus uhdei* (Wenz.) Lingelsh.	Ash tree	X	X	X
13	*Jacaranda mimosifolia* D.Don	Blue jacaranda	X	X	X
14	*Lagerstroemia indica* L.	Crepe myrtle			X
15	*Ligustrum lucidum* W.T.Aiton	Glossy privet	X	X	X
16	*Liquidambar styraciflua* L.	Sweet gum			X
17	*Malus domestica* Borkh.	Apple	X		X
18	*Melia azedarach* L.	Chinaberry tree			X
19	*Morus nigra* L.	Black mulberry	X		X
20	*Pinus spp.*	Pine	X	X	X
21	*Populus alba* L.	White poplar	X	X	
22	*Populus* x *canadensis* Moench	Canadian poplar	X	X	
23	*Neltuma laevigata* (Willd.) M.C.Johnst.	Mesquite	X	X	X
24	*Prunus cerasifera* Ehrh.	Cherry plum			X
25	*Prunus persica* (L.) Batsch	Peach			X
26	*Prunus x yedoensis* Matsum	Cherry	X		
27	*Quercus virginiana* Mill.	South oak		X	
28	*Schinus molle* L.	Peruvian peppertree	X	X	X
29	*Schinus terebinthifolia* Raddi	brazilian peppertr	X	X	X

30	*Taxodium huegelii* C.Lawson	Moctezuma blad cypress	X		
31	*Ulmus pumila* L.	Siberian elm			X
32	*Vachellia schaffneri (S.Watson) Seigler & Ebinger*	Huisache	X	X	X