



Original article

Twelve-year change in tree diversity and spatial segregation in the Mediterranean city of Santiago, Chile

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ABSTRACT

Tree diversity is one of the most important components of urban ecosystems, because it provides multiple ecological benefits and contributes to human well-being. However, the distribution of urban trees may be spatially segregated and change over time. To provide insights for a better distribution of tree diversity in a socially segregated city, we evaluated spatial segregation in the abundance and diversity of trees by socioeconomic group and their change over a 12-year period in Santiago, Chile. Two hundred vegetation plots were sampled across Santiago in 2002 and 2014. We found that overall abundance and diversity of urban trees for the entire city were stable over 12 years, whereas species richness and abundance of native tree species increased. There was segregation in tree species richness and abundance by socioeconomic group, with wealthier areas having more species and greater abundance of trees (for all tree species and native species) than poorer ones. Tree community composition and structure varied with socioeconomic group, but we found no evidence of increased homogenization of the urban forest in that 12 years. Our findings revealed that although tree diversity and abundance for the entire city did not change in our 12-year period, there were important inequities in abundance and diversity of urban trees by socioeconomic group. Given that 43% of homes in Santiago are in the lower socioeconomic areas, our study highlights the importance of targeting tree planting, maintenance and educational programs in these areas to reduce inequalities in the distribution of trees.

1. Introduction

Plant diversity, especially of trees, is one of the most important components of urban ecosystems. It provides multiple ecological benefits and contributes directly to the value of public life and human well-being (Dobbs et al., 2014; Scopelliti et al., 2016). More diverse urban forests may contribute to the optimization of multiple ecosystem services mainly because some species are better than others in providing a particular service (Morgenroth et al., 2016 and references therein). In addition, a more diverse urban forest can increase its resilience to pests and disease (Santamour, 1990). It can also contribute to local biodiversity conservation by preserving native tree species in urban environments and providing habitat for local animal species (Alvey, 2006; Ikin et al., 2015; Villaseñor et al., 2017).

Urban vegetation is a human-created mix of native and alien species, combined in different proportions and spatially varying throughout the city according to local habitats and sociocultural decisions (Kinzig et al., 2005; Avolio et al., 2015). Hope et al. (2003) suggested that a functional relationship may link human wealth and plant diversity in urban ecosystems (the “luxury effect”). In urban

landscapes, the abundance and diversity of plants are positively related with neighborhood income. Higher income neighborhoods tend to have greater abundance and more diverse public and private green spaces, whereas the poorest neighborhoods tend to have lower quantities and less diverse ones (Kirkpatrick et al., 2011; Clarke et al., 2013). This segregation in the spatial distribution of vegetation may be not only present in large cities in more developed countries, but also in medium-sized cities in less developed countries (e.g., cities in Latin America, de la Maza et al., 2002; Borsdorf and Hidalgo, 2010). Because trees provide contact with nature for urban-dwellers, which may influence their well-being (Scopelliti et al., 2016), it is important to plan for sustainable cities that provide tree benefits to all their residents.

To plan for sustainable cities, it is important to understand how the composition and structure of urban vegetation changes in space and over time. Urbanization may cause biotic homogenization (i.e., increasing similarity of species composition, decreasing beta diversity over time), where a few human-selected species become widespread (McKinney, 2006; Qian et al., 2016). The preference for alien species in tree planting programs, due to high growth rates or esthetic criteria, is one of the main consequences of urbanization and may increase biotic

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homogenization across cities around the world (Gong et al., 2013).

In this study we provide insights for a better distribution of trees in a socially segregated city. For this, we present relevant empirical evidence on the abundance and diversity of trees in Santiago, Chile. Santiago is located in a biodiversity hotspot characterized by a high level of plant endemism, but where most of the native vegetation has been modified for anthropogenic land use (Myers et al., 2000; Pauchard and Barbosa, 2013). This city has a strong socio-spatial segregation and an uneven distribution of urban green space (Romero et al., 2012; de la Barrera et al., 2016).

Our overall aim was to evaluate differences in the abundance and diversity of trees among socioeconomic areas (high, middle and low) and their changes over a 12-year period (2002–2014) in Santiago, Chile. For this we evaluated differences in diversity and abundance of all trees, native and exotic species, among socioeconomic groups and years. We also investigated whether urban forests in wealthier areas have greater cumulative species richness than in other socioeconomic areas. Finally, we explored whether biotic homogenization of Santiago's urban forest has occurred in the 12-year period.

In contrast to most urban tree diversity research, which has been biased towards green spaces and public land, here we present detailed tree surveys on both public and private land which provides a better representation of tree diversity in the city. By studying the Mediterranean urban forest of Santiago, Chile, we also address the geographic bias in tree-related research, where most scientific information has been generated in North America, Europe, and Australia (Roy et al., 2012; Nitoslawski et al., 2016). This study also comprises a relatively long temporal scale (12 years) and large spatial scale (entire city). Our findings provide empirical evidence to report on urban forestry practices in the region and contribute to a growing body of knowledge on tree diversity in urban environments.

2. Methods

2.1. Study area

The study area comprises the urban extent of Santiago, the capital of Chile, which encompasses an area of $\sim 967 \text{ km}^2$ (Fig. 1). The city has around 7 million inhabitants and integrates multiple land uses, land covers and tenures. According to the Köppen climate classification system, Santiago is located in warm-summer Mediterranean climate zone ($33^{\circ}27' \text{ S}$ - $70^{\circ}41' \text{ W}$), characterized by dry and warm summers (Peel et al., 2007). Monthly average for the warmest month (January) is $\sim 20^{\circ} \text{ C}$, and for the coolest month is $\sim 8^{\circ} \text{ C}$ (July); while mean rainfall is $\sim 312.5 \text{ mm}$ per year. The elevation of the city varies from 400 to 900 m (average: 540 m). The eastern part of the city has higher elevation because it is located on the Andean piedmont and is mainly covered by shrublands. The western part of the city, once an *Acacia* spp. and grass-dominated alluvial plain, has been transformed to agricultural and urban land cover with little of its original vegetation cover remaining (Romero et al., 2012).

2.2. Sampling design

Two-hundred circular plots with an 11 m radius (0.038 ha) were allocated across the study area using a stratified random approach (criteria and methods described in detail by Escobedo et al., 2006) (Fig. 1). These plots were first measured in 2002 and then re-measured in 2014 using the same protocols. In both years, plot surveys were performed during the austral summer. Detailed vegetation data were recorded in each plot, where every tree ($> 2.5 \text{ cm}$ diameter at breast height, DBH) was identified at species level and measured to record its size and frequency.

2.3. Socioeconomic groups

A socioeconomic group was assigned to each plot in a second phase by crossing the sampling location with a socioeconomic map (ADIMARK, 2012). Adimark's map (2012) takes data from 2002 national census and assigns a socioeconomic status to a home based on head of household education level and 10 selected goods (e.g. shower, fridge, computer, internet, car). A socioeconomic group is later assigned to a squared block according to the dominant status within the block. Here we summarize three socioeconomic groups:

High (ABC1): Head of household usually has complete university education. They own on average 9.2 goods of a maximum of 10 selected goods. High annual income. This segment represents 11.3% of the homes in Santiago.

Middle (C2C3): Head of household has secondary or tertiary education. On average, they commonly own 5–8 of selected goods. Medium annual income. This segment represents 45.7% of the homes in Santiago.

Low (DE): Head of household has incomplete primary or secondary education. They commonly own < 6 of selected goods. Low income. This segment represents 43% of the homes in Santiago.

This map has higher spatial resolution ($30 \text{ m} \times 30 \text{ m}$) than those used previously in our study area (e.g. municipality-level, de la Maza et al., 2002; Escobedo et al., 2016).

Plots located on sites where a socioeconomic group was not available (e.g. piedmont, shrublands, agricultural lands) were not considered in our analysis. Therefore, our analysis included 168 and 167 plots measured in 2002 and 2014, respectively.

2.4. Statistical analysis

We first evaluated tree diversity and abundance in our sampling plots between years and socioeconomic groups. We calculated tree species richness and abundance, native species richness and abundance, and exotic species richness and abundance per sampling plot. To evaluate differences in tree diversity and abundance between years and socioeconomic groups within years, we first plotted our response variables by year and socioeconomic group using three levels: high (ABC1), middle (C2C3), and low (DE). Because our data did not comply with the assumptions of parametric tests, we used Kruskal-Wallis rank-sum tests to test for differences between years and socioeconomic groups in R (R Core Team, 2015). When significant differences were found by socioeconomic groups, we calculated pairwise multiple comparisons between group levels to detect the levels that differed (Nemenyi-tests with Chi-squared approximation) (Pohler, 2014).

We also calculated two commonly used diversity indices for each year and socioeconomic group:

(1) Shannon-Weaver

$$H' = -\sum_{i=1}^S p_i \ln p_i$$

(2) Simpson's Index of Diversity

$$D_1 = 1 - \sum_{i=1}^S p_i^2$$

Where p_i is the proportional abundance of species i , and S is the number of species, so $\sum_{i=1}^S p_i = 1$, and \ln is the natural logarithm. For Simpson's Index of Diversity, values range between 0 and 1, where the greater the value, the greater the sample diversity. This index gives more weight to abundant species, so rare species cause small changes to the index.

We used randomization tests to compare differences in cumulative species richness – for both total and native species – between years and to compare differences between high socioeconomic group and middle and low groups within each year. Randomization tests were performed

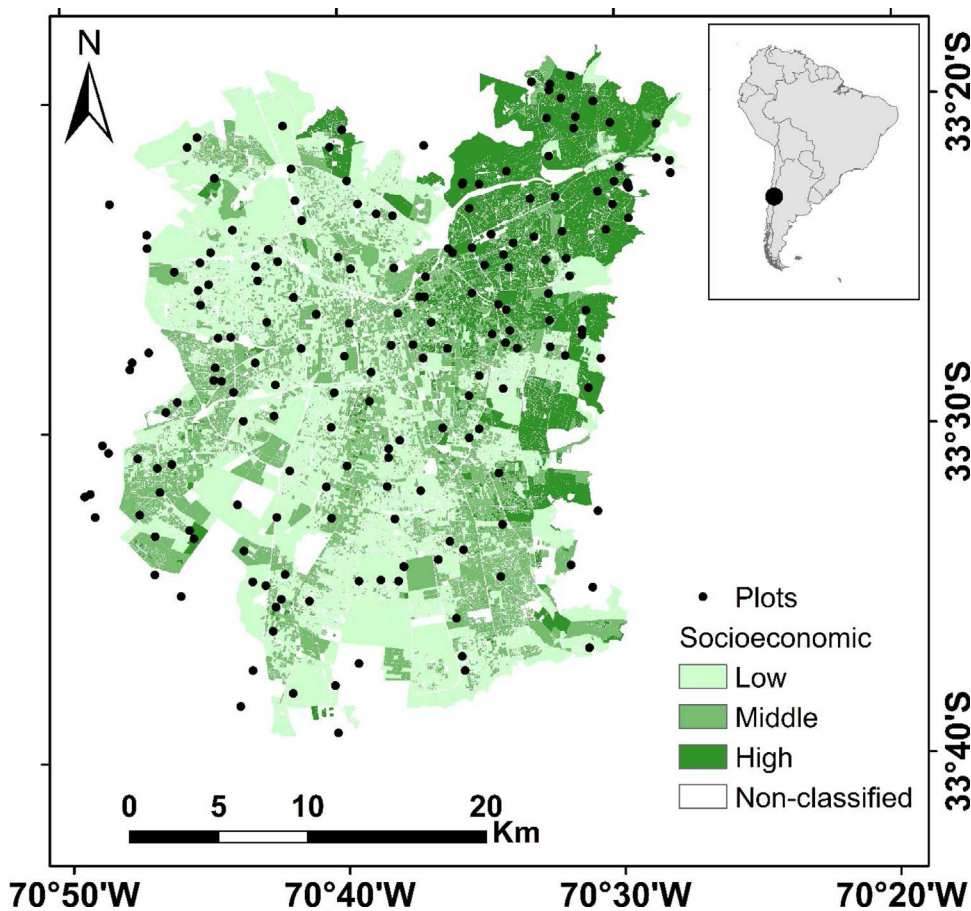


Fig. 1. Spatial location of sampling plots and socioeconomic groups in Santiago, Chile.

using the *rich* package in R (with 999 randomizations) (Rossi, 2011). In addition, for each year we built rarefied species accumulation curves of total tree species and native tree species for the three socioeconomic groups using the *vegan* package in R (Oksanen et al., 2016). We used the sample-based rarefaction method (Mao Tau estimate, Colwell et al., 2012) to estimate the number of species with increasing sample size. To find species richness by sampling individuals instead of sites, we used the individual-based rarefaction method (Oksanen et al., 2016).

To evaluate differences in species composition and their relative abundance we used a multivariate ordination of beta diversity with principal coordinates analysis (PCoA). For this, we first removed empty sites from our community data. We calculated two measures of ecological dissimilarity: Sørensen and Bray-Curtis. Sørensen is a widely used ecological measure of compositional dissimilarity based on presence-absence. Bray-Curtis is a dissimilarity measure that includes relative abundance. Prior to calculating a Bray-Curtis dissimilarity matrix between sampling plots, we standardized each site to the same abundance by dividing cell abundance by the total abundance for each site (leading to species proportions) (Kindt and Coe, 2005).

We used non-parametric permutational multivariate analysis of variance (PERMANOVA) (Anderson, 2001) to test for differences in beta diversity by socioeconomic group and year (beta diversity was represented by Sørensen and Bray-Curtis dissimilarity measures). A PERMANOVA partitions dissimilarities for the sources of variation (socioeconomic group and year), and performs permutation tests to detect statistical significances. Because differences in PERMANOVA may be caused by dispersion effects, we used a permutational analysis of multivariate dispersions (PERMDISP) (Anderson, 2006), which tests whether classes differ in their within-group dispersion (distance to group median). This test permutes model residuals to generate a permutation distribution of F under the null hypothesis of no difference in

dispersion between groups. If significant differences were found, we performed pairwise comparisons. PERMANOVA and PERMDISP tests were performed with the *adonis* and *betadisper* functions, respectively, from the *vegan* package (Oksanen et al., 2016) in R.

3. Results

Exotic trees dominated the urban forest. Of 494 individual trees recorded in 2002, 96% were exotic species and only 4% were native to the country (Table 1, Appendix Tables 1–2). Of 488 individual trees recorded in 2014, 86% were exotic species and 14% were native to the country (Table 1, Appendix Tables 1–2). Of the 137 different tree species recorded, 126 (92%) were exotic species and only 11 (8%) were native species. *Prunus cerasifera* and *Robinia pseudoacacia* were the most abundant species in 2002, comprising 20% of the total abundance of trees. In 2014, three species (*Prunus cerasifera*, *Liquidambar styraciflua* and *Robinia pseudoacacia*) comprised 20% of the abundance of sampled trees (Appendix Table 2).

Similar species richness and abundance per plot were found between 2002 and 2014 ($P = 0.58$ and $P = 0.94$, respectively) and among socioeconomic groups in 2002 ($P = 0.4$ and $P = 0.6$, respectively). However, for species richness we found close to marginal differences among socioeconomic groups in 2014 ($P = 0.08$), where plots in high socioeconomic areas had more tree species than plots in low socioeconomic ones (Table 2). In addition, plots in high socioeconomic areas had significantly more trees than plots in low socioeconomic areas in 2014 ($P = 0.03$).

Native species richness and abundance was higher in plots surveyed in 2014 compared to plots surveyed in 2002 ($P = 0.04$ and $P = 0.03$, respectively). Plots in low socioeconomic areas had significantly less native species richness and abundance than plots in high socioeconomic

Table 1

Number of plots and trees (native and exotic) recorded by socioeconomic group and year in Santiago, Chile. For native and exotic trees, the number of individuals and their percentage contribution to the total are shown. Note that these plots are a subset of the original 200 plots and represent those plots where a socioeconomic group was identified according to Adimark's map (see Methods).

	2002			2014			Relative percent change ^a					
	Total	Socioeconomic group		Total	Socioeconomic group		Total	Socioeconomic group				
		High	Middle		Low	High		Middle	Low			
Number of plots	168	56	38	74	167	54	51	62	-1%	-4%	34%	-16%
Number of plots with trees	108	36	26	46	111	42	31	38	3%	17%	19%	-17%
No. of trees	494	171	125	198	488	202	142	144	-1%	18%	14%	-27%
- No. of native trees	19 (4%)	15	3	1	66 (14%)	32	25	9	247%	113%	733%	800%
- No. of exotic trees	475 (96%)	156	122	197	422 (86%)	170	117	135	-11%	9%	-4%	-31%

^a Relative percent change = 100*(value in 2014-value in 2002)/value in 2002.

areas in 2002 ($P < 0.004$) but these differences become non-significant in 2014 ($P \geq 0.08$, Table 2). Exotic species richness and abundance in plots were similar between years and socioeconomic groups ($P > 0.2$).

Shannon-Weaver and Simpson's Index of Diversity tended to increase with socioeconomic status. They also exhibited similar values between years (Table 3).

There were no differences in the cumulative number of tree species recorded in 2002 compared with 2014 for the entire city (106 and 105 species, respectively, $P = 0.47$, Table 4A). However, higher socioeconomic areas had higher cumulative species richness than lower socioeconomic ones. Difference in cumulative species richness between high socioeconomic areas (62 species) and middle socioeconomic areas (49 species) was non-significant in 2002 ($P = 0.54$), because the middle socioeconomic areas had fewer sampled plots (Fig. 2). The difference in cumulative species richness between high socioeconomic areas (62 species) and low socioeconomic areas (53 species) was significant in 2002 ($P = 0.03$, Fig. 2). These differences were more significant in 2014 (64 vs. 45 species, $P = 0.005$, Fig. 2). Individual-based species accumulation curves revealed that increased differences in species richness between high and low socioeconomic areas in 2014 compared to 2002 may be due to a smaller number of trees recorded in 2014 in lower socioeconomic areas (Fig. 3).

We found significant differences in the cumulative number of native tree species recorded in 2002 compared with 2014 for the entire city (6 and 11 species, respectively, $P = 0.017$, Table 4B). When comparing the difference in cumulative species richness of native tree species in 2002, high socioeconomic areas had significantly more native tree species than middle (6 vs. 1 species, $P = 0.014$, Table 4B) and low socioeconomic areas (6 vs. 1 species, $P = 0.002$, Figs. 2 and 3). In 2014, the differences between high and middle socioeconomic areas decreased and were non-significant (9 vs. 7 species, $P > 0.34$); however, the higher socioeconomic areas continue to have significantly more native species than the lower socioeconomic areas (9 vs. 4 species, $P = 0.01$, Fig. 2). Individual-based species accumulation curves for native species revealed that the low number of native species in the

Table 2

Mean (range) of diversity variables per plot (0.04 ha) by socioeconomic group and year recorded in Santiago, Chile. P shows statistical significance between socioeconomic groups from non-parametric tests (Kruskal-Wallis). Different letters indicate groups that differed according to post hoc pairwise comparisons using Nemenyi tests (see Methods). P -year shows statistical significance between years (2002 vs. 2014). Significant and marginally significant P values ($P < 0.05$ and $P \leq 0.1$, respectively) are indicated in bold.

Variables	2002				P	2014				P year	
	High	Medium	Low			High	Medium	Low	P		
Species richness	1.89 (0-6)	2.32 (0-10)	1.55 (0-6)		0.4	2.28 (0-6)	1.96 (0-9)	1.6 (0-7)		0.08	0.58
- Native species richness	0.18 (0-1) ^a	0.08 (0-1) ^{a,b}	0.01 (0-1) ^b		0.004	0.28 (0-2)	0.2 (0-3)	0.11 (0-2)		0.1	0.04
- Exotic species richness	1.71 (0-6)	2.24 (0-10)	1.54 (0-6)		0.49	2 (0-6)	1.77 (0-9)	1.48 (0-6)		0.2	0.91
Abundance of trees	2.5 (0-15)	3 (0-13)	1.5 (0-13)		0.62	3(0-13) ^a	2 (0-22) ^{a,b}	2 (0-10) ^b		0.03	0.94
- Abundance of native trees	0.27 (0-3) ^a	0.08 (0-1) ^{a,b}	0.01 (0-1) ^b		0.003	0.59 (0-7)	0.49 (0-15)	0.15 (0-3)		0.08	0.03
- Abundance of exotic trees	2.79 (0-15)	3.21 (0-13)	2.66 (0-13)		0.73	3.15 (0-13)	2.29 (0-12)	2.18 (0-10)		0.21	0.42

Table 3

Values of diversity indices by year and socioeconomic area in Santiago, Chile.

Diversity index	Year	Socioeconomic group		
		High	Middle	Low
Shannon-Weaver	2002	3.74	3.62	3.26
	2014	3.71	3.65	3.27
Simpson's Index of Diversity	2002	0.97	0.97	0.92
	2014	0.96	0.96	0.94

Table 4

Comparisons of (A) cumulative species richness and (B) cumulative native species richness between years and between socioeconomic groups within years. For socioeconomic groups, we compared wealthier socioeconomic level (high) with lower socioeconomic levels (middle and low). Community 1 and 2 refer to the observed values for the first and second group compared, respectively. P values from randomization tests (number of randomizations per test = 999). Significant P values are indicated in bold.

	2002	2014		P	
		2002-2014	High-Middle		High-Low
<i>(A) Cumulative species richness</i>					
Community 1	106	62	62	64	64
Community 2	105	49	53	55	45
Community 1-Community 2	1	13	9	9	19
P	0.47	0.54	0.03	0.26	0.005
<i>(B) Cumulative native species richness</i>					
Community 1	6	6	6	9	9
Community 2	11	1	1	7	4
Community 1-Community 2	-5	5	5	2	5
P	0.017	0.014	0.002	0.34	0.01

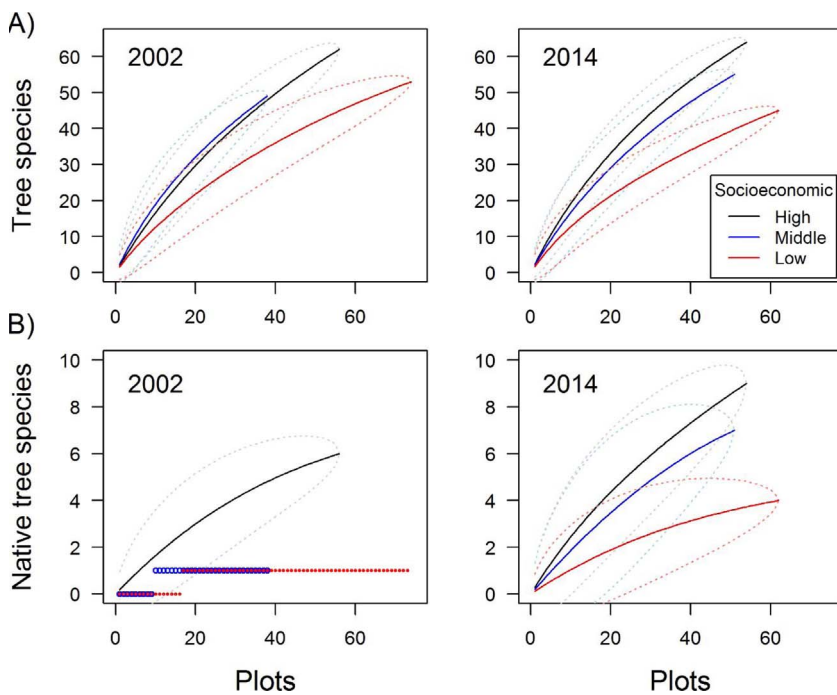


Fig. 2. Sample-based species accumulation curves for (A) all tree species and (B) native species recorded during vegetation surveys in Santiago. Missing accumulation species curves describing middle and low socioeconomic groups (blue and red line, respectively) for native species in 2002 is due to the recording of only one species of native tree (see Table 1). (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

middle and lower socioeconomic areas is mainly due to the small number of native individuals recorded in these areas (Fig. 3).

When evaluating dissimilarities between sites in their species composition and relative abundance, we found for both Sørensen and Bray-Curtis distances significant differences due to socioeconomic group ($P = 0.001$) but not due to year ($P > 0.5$) or their interaction ($P > 0.8$, Table 5). The effect of socioeconomic status seems to be primarily due to differences in the within-class dispersion (Sørensen, $P = 0.06$; Bray-Curtis, $P = 0.04$; Table 6). Lower dispersion was found in high socioeconomic areas (thus plots were more homogeneous) than in lower socioeconomic areas (Fig. 4). Higher dispersion in the multivariate space for low socioeconomic areas was mainly due to *Robinia pseudoacacia*. This species was associated with the second axis of the

Table 5

Results of permutational multivariate analysis of variance for urban trees indicating the effect of socioeconomic group and year on multivariate dissimilarities according to Sørensen and Bray-Curtis distances. Significant P values are indicated in bold.

Sources of variation	df	Sørensen			Bray-Curtis		
		SS	F	P	SS	F	P
Socioeconomic group	2	2.66	2.95	0.001	2.51	2.73	0.001
Year	1	0.41	0.91	0.59	0.44	0.96	0.51
Socioeconomic group*Year	2	0.73	0.81	0.84	0.72	0.78	0.90
Residuals	213	95.82	0.45		97.71		
Total	218	99.61			101.38		

Significance of the *pseudo F-ratio* was tested with permutation test (999 permutations).

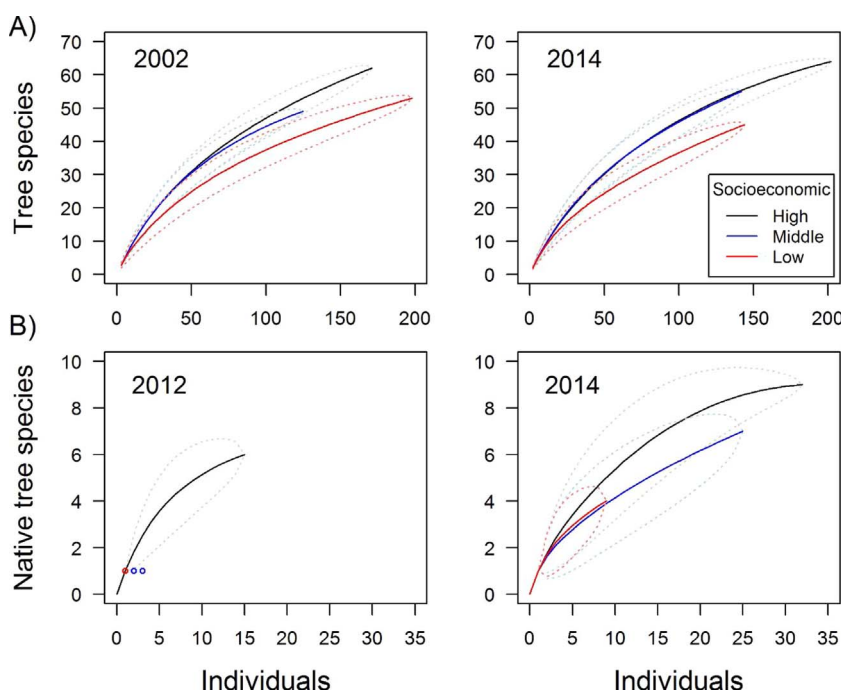


Fig. 3. Individual-based species accumulation curves for (A) all tree species and (B) native species recorded during vegetation surveys in Santiago. Missing rarefied species accumulation curves describing middle and low socioeconomic groups (blue and red line) for native species in 2002 is due to the recording of only one native species. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

Table 6

Results of tests for homogeneity of multivariate dispersions for urban trees based on multivariate dissimilarities according to Sørensen and Bray-Curtis distances. Significant and marginally significant *P* values (< 0.05 and < 0.1 , respectively) are indicated in bold.

	Sørensen		Bray-Curtis	
	F	P	F	P
Socioeconomic group	2.66	0.06	3.04	0.04
Year	0.027	0.86	0.01	0.91

Where there was a statistically significant or marginally significant overall *F*-ratio comparing groups ($P < 0.1$, 999 permutations), pairwise comparisons were done. Different letters indicate groups that were significantly different ($P < 0.05$).

PCoA, and it was more frequent in the lower socioeconomic areas. In contrast to the effect of socioeconomic group, we did not find any significant effect on multivariate dispersions due to year ($P > 0.8$, Fig. 4).

4. Discussion

A better understanding of temporal and spatial patterns of tree diversity is crucial to reporting on urban forestry practices. Here we evaluated spatial segregation and the 12-year change in tree diversity and abundance in the Mediterranean urban forest of Santiago, Chile. We found that species richness and abundance of native species increased, but overall abundance and diversity of urban trees for the entire city were stable over 12 years. Inequalities in tree species richness and abundance by socioeconomic status continue, with high socioeconomic areas having more species and abundance of trees (for all tree species and native species). Although tree species composition between plots in higher socioeconomic areas tended to be less dissimilar than between plots in lower socioeconomic areas, we did not find evidence for an increased homogenization of Santiago’s urban forest in the 12 year period.

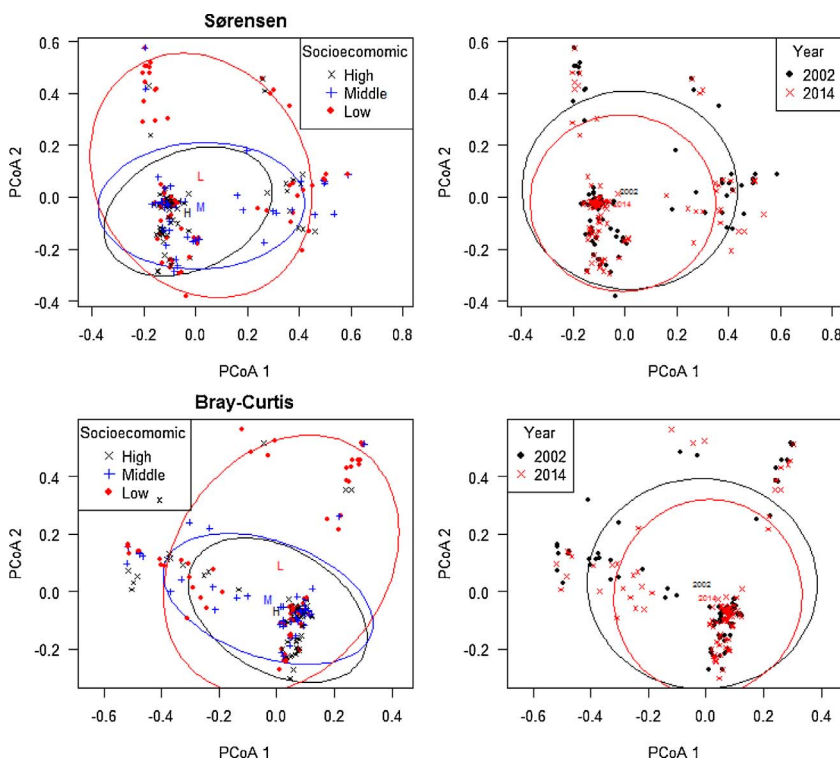


Fig. 4. Principal coordinates analysis ordination of Sørensen and Bray-Curtis dissimilarities between sampling plots. Plots show the effect of socioeconomic status and year on beta diversity. The ellipse encloses 80% of the dispersion of groups from their respective group centroid.

4.1. Santiago’s urban forest

The proportion of native and exotic species in urban forests varies greatly among cities. While some cities have an urban forest primarily comprised of native species (e.g. $> 90\%$ of trees and shrubs recorded in Changchun, China, are native species) (Zhang et al., 2016), Santiago’s urban forest is dominated by exotic species with a very low proportion of native species. More than 90% of tree species in Santiago are exotic to the country, where exotic trees comprised 96% and 86% of the total number of trees measured in 2002 and 2014, respectively. A low proportion of native species was also reported in the urban forest of southern California (USA), where Avolio et al. (2015) found 10 fold more exotic species than native species. The dominance of exotic over native species likely decreases local ecological integrity, because it may promote the invasion of alien species and the loss of endemic species (Alvey, 2006). This is particularly important in Latin America, where rapid urbanization threatens biodiversity hotspots (Pauchard and Barbosa, 2013).

Tree species in cities located in areas colonized by Europeans have traditionally been chosen from a European species pool (e.g. cities in Canada, Australia, New Zealand, reviewed by Nitoslawski et al., 2016). In Santiago, the overwhelming prevalence of exotic species reflects a strong influence of not only European design in urban forestry and greening, but also species from North America, Oceania and Asia (Alvarado et al., 2013; Fischer et al., 2016). For instance, the three most abundant species in Santiago had different origin: *Prunus cerasifera* (Europe and Asia), *Liquidambar styraciflua* (North and Central America) and *Robinia pseudoacacia* (North America). These species are also among the most frequent species of urban trees in other cities in Chile, such as Concepción (Pauchard et al., 2006). This preference for exotic species from the Northern Hemisphere can also be found in other cities with semi-arid climates of South America. For example, some of the most common urban trees in Mendoza, Argentina, correspond to species from Asia (e.g. *Morus alba*), Europe (e.g. *Fraxinus excelsior*) and North America (e.g. *Fraxinus Americana*) (Martinez et al., 2013; Dirección de Paseos, 2017).

Past urban forestation programs have preferred exotic species over

native ones. This preference for exotic species is commonly associated with rapid growth rates, aesthetics, nursery stock, and a perception of higher tolerance to urban stress and pests (Alvarado et al., 2013). It has also been argued that exotic species can contribute to fulfill ecosystem services in regions with few native tree species suitable for urban environments. For instance, only one native tree species would be a good candidate for providing shade in inner-city paved areas in Scandinavia; suggesting the need for exotic species to help mitigate the urban heat island effect (Sjöman et al., 2016).

However, growing evidence suggests that planning for an urban forest that favors native species might increase resilience to pests and disease, preserve local tree species, and provide habitat for native species in the urban environment (Nitoslawski et al., 2016). A common recommendation to increase protection against pest and disease outbreaks is that the urban forest should be comprised of plants < 10% of any species (Santamour, 1990). In Santiago, the most dominant species represented < 10% of the total abundance of trees. However, 20% of urban trees were represented by only 2–3 exotic tree species which makes 1/5 of the urban forest susceptible to pests affecting only a couple of tree species. Given that the most frequent native species (*Acacia caven*) represents < 4% of trees in any year, it is advisable to plan for an urban forest that favors native species.

In addition, favoring native species in Santiago may also help improve air quality. This is because many exotic tree species lose their leaves during winter and produce large amounts of potentially detrimental compounds (e.g. biogenic volatile organic compounds, BVOC), which are likely to negatively impact air quality and human health (Prendez, 2016). Furthermore, in water-restricted environments, such as arid and Mediterranean environments, exotic species usually incur high irrigation costs whereas preferring native tree species contribute to reducing irrigation costs and the detrimental effects of water deficit on tree growth (Alvarado et al., 2013; Martínez et al., 2013). This is particularly important in cities in the semi-arid environments of South America, where there is a loss of tree vitality due to insufficient maintenance (e.g. irrigation, pruning) and damaged irrigation systems (Breuste, 2013).

4.2. 12-year change in tree diversity and abundance

Our findings reveal that Santiago in 2014 had similar abundance and diversity of trees than 12 years earlier. This finding is consistent with previous analyses that show tree mortality, ingrowth and biomass in this city have remained stable in recent years (Escobedo et al., 2016). Although urban forestation programs have been developed by national and local authorities (e.g., Piñera, 2006; CONAF, 2016), our findings show that these programs may have contributed to maintain tree cover in the city, but they have not increased tree abundance, species richness or overall diversity.

Although tree abundance and diversity have been stable over 12 years, we found an increase in species richness and abundance of native trees. This increase in native species is likely due to the promotion of native species over exotic species by The Chilean Forestry Service (CONAF). CONAF plays an important role in policy and on-the-ground urban forestation programs, where this service not only supports local communities in technical matters, but also favors the use of native species for biodiversity conservation (CONAF, 2016). The importance of native species in tree planting programs has been increasingly highlighted for the restoration of native ecosystems in urban landscapes. In addition to Chile, explicit policies to increase the presence of native species have been adopted elsewhere (e.g., North America, Almas and Conway, 2016). Although it has been argued that this native-driven approach may affect urban ecosystem resilience in regions with extreme environmental conditions (Sjöman et al., 2016), the overall goal is to increase ecological integrity of local environments (Almas and Conway, 2016; Johnson and Handel, 2016).

4.3. Spatial segregation by socioeconomic group

Our findings reveal that wealthier areas have a higher density of trees and more tree species than poorer areas, and these differences seem to increase over time. The relationship between wealth and tree diversity was reported at municipality level in Santiago by de la Maza et al. (2002), and our study found that this pattern is still present at the neighborhood level (square block). Increased segregation with socioeconomic status is likely a result of the gain in tree cover in high socioeconomic areas and the loss of tree cover in lower socioeconomic areas (Escobedo et al., 2016). The increase in tree cover inequality over time has been reported elsewhere, where income level can be used as a reasonable predictor of future canopy cover (e.g., Australia, Krafft and Fryd, 2016).

While differences in overall species richness and abundance between higher and lower socioeconomic areas have increased over the last decade, differences in native species richness and abundance decreased. However, neighborhoods of lower socioeconomic status still have very few native trees.

When countries do not have species diversity policies for urban forestry, tree species on both public and private property depend on personal preferences and the priorities of practitioners, such as arborists, developers, landscape architects and residents, as well as local plant stock (Nitoslawski et al., 2016; Qian et al., 2016). Urban forests are influenced by the behavior of numerous households who decide what to plant on their property and on the public land outside it. Household behavior is particularly important in the composition and structure of Santiago's urban forest because 45%–70% of trees in Santiago are on private property (Escobedo et al., 2006). Household behavior towards tree planting and removal can be influenced by socioeconomic status. Groups with higher education and income are more likely to perceive trees as a positive contribution to the urban landscape and to plant trees for esthetic and ecological benefits (Nitoslawski et al., 2016). It is likely that these attitudes cause differences among the socioeconomic areas in our study – where socioeconomic groups were defined by education and purchasing power (reflected by goods or services in a home). If the differing attitudes towards tree benefits and tree planting continue, spatial segregation in tree diversity and abundance are likely to increase over time.

In addition to household planting, tree species composition may be influenced by municipal policies which may favor wealthier groups for public plantings (Nitoslawski et al., 2016). Escobedo et al. (2006) found that wealthier municipalities in Santiago had more budget and tended to spend more money per tree than poorer ones. Although targeted tree-planting programs have been implemented in poor neighborhoods in Santiago, they do not appear to effectively reduce differences in the spatial distribution of trees.

4.4. Biotic homogenization of the urban forest

Biotic homogenization may be found across urban areas and over time, mainly due to the establishment of common alien species in urban areas (McKinney, 2006; Gong et al., 2013; Qian et al., 2016) – although alien species may initially increase biodiversity, contributing to differentiation (Lososová et al., 2012). Our findings evidenced that tree species composition and relative abundance were more homogeneous in plots located in higher socioeconomic areas than in lower socioeconomic ones. Some plots in lower socioeconomic areas were more dissimilar to the rest of the plots due to the presence of *Robinia pseudoacacia*, a species more common and abundant in neighborhoods of lower socioeconomic status.

However, we found no evidence of increased biotic homogenization of Santiago's urban forest over 12 years. Our two measures of ecological dissimilarity, Sørensen and Bray-Curtis, did not show significant changes in dissimilarity of species composition and relative abundance between vegetation plots (beta diversity) between 2002 and 2014. This

finding may be a result of our time frame. A decade may be too short a time to detect taxonomic homogenization, differentiation or shifts in tree community composition in our study area with the multivariate methods we used (PCoA).

When comparing the compositional similarity of urban forests among 38 cities from four continents (North America, Asia, Europe and South America), Yang et al. (2015) found that Santiago's urban forest was more similar to cities with Mediterranean climates than to other biomes. The composition of Santiago's urban forest is similar to the urban forest of California, USA (e.g. San Francisco and Los Angeles) and Barcelona, Spain (Yang et al., 2015). Some tree species contributing to the similarity between urban forests from southern California (data from Avolio et al., 2015) and Santiago may include *Liquidambar styraciflua*, *Prunus cerasifera*, *Ligustrum japonicum*, and *Platanus x acerifolia*. Some tree species likely contributing to the similarity between urban forests from Barcelona (data from Chaparro and Terradas, 2009) and Santiago are *Robinia pseudoacacia*, *Platanus x acerifolia*, *Acer negundo*, *Acacia dealbata*, and *Eriobotrya japonica*. *Schinus molle*, a tree species originally from arid zones in South America, also contributes to this similarity. *Schinus molle* is abundant in Mediterranean cities and has become widely naturalized in other regions (Richardson and Rejmanek, 2011).

5. Conclusion

Santiago is located in an area of high plant endemism; however, Santiago's urban forest is primarily composed of exotic species: more than 90% of tree species are exotic to the country, comprising 84–96% of the total number of trees. Public tree planting programs likely increased native trees from 4% to 14% for the period 2002–2014, and reduced differences in native species richness and abundance among areas of disparate socioeconomic status.

Despite the tree planting programs, Santiago exhibits a strong segregation in tree abundance and diversity. Wealthier areas have higher density of trees and more tree species than poorer areas. The large representation of poorer socioeconomic groups in Santiago (43% of total homes, ADIMARK, 2012) highlights the need for targeted tree planting, maintenance and educational programs in those areas to reduce inequalities in the spatial distribution of trees.

Overall tree abundance, diversity and community composition and structure for the city remained stable over 2002–2014, revealing a lack of evidence for biotic homogenization of Santiago's urban forest in 12 years.

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Appendix A. Supplementary data

Supplementary data associated with this article can be found, in the online version, at <http://dx.doi.org/10.1016/j.ufug.2017.10.017>.

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