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Abundance and physiognomic characteristics of woody natives in a sub-urban talar forest remnant extensively invaded by exotic vegetation

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ABSTRACT

Introduction: The talares are small xerophytic forests dominated by the trees *Celtis tala* and *Scutia buxifolia*, which are characteristic of the pampas in the Northeast of Buenos Aires Province, Argentina. Talares coexist with areas facing significant urban and agricultural pressures, leading to their impoverishment, fragmentation, and even local extinction.

Objective: To characterize a resilient talar patch within a suburban forest invaded by exotic vegetation to identify the patterns and processes currently affecting native woody species.

Methods: The forest (55 ha) was divided into seven sampling sites. We assessed the abundance and physiognomic variables of native woody species through random searches (approach 1). For exotic woody species, we used a combination of transects and grids within the forest and on adjacent streets (approach 2). The analyses included descriptive statistics, index calculations, and comparisons through confidence interval establishment and Chi-square tests with Yates' correction.

Results: Total sampling time per team member was 118.5 h for approach 1 and 48 h for approach 2. Two native species dominate the studied talar patch: *C. tala* (88 %) and *S. buxifolia* (11 %), while other native woody species were extremely rare (1 %). Exotic woody species were dominant structurally and numerically. We observed variations of *C. tala* and *S. buxifolia* height between forest stands, most common under 3 m in height. The Mantle Index indicated light competition due to shading caused by fast-growing, large exotic species. Damage due to falling eucalyptus branches was observed only in *C. tala*, although at a low percentage. *S. buxifolia* had 20 % and *C. tala* 15 % of individuals with a shrub-like form with some cases being variable among forest stands. We propose several manageent strategies aimed at favoring native species over exotics.

Conclusion: The studied forest is a hybrid ecosystem with neo-ecosystem traits, which implies a series of conservation problems for the remaining native woody species present. It is highly valuable for the conservation of talares, particularly for one of its most emblematic species, *S. buxifolia*, which is virtually absent outside the forest area.

Key words: adaptive management; forest resilience; hybrid ecosystems; suburban forests; habitat loss; woody natives.

RESUMEN

Abundancia y características fisonómicas de leñosas nativas de un talar suburbano remanente extensamente invadido por exóticas

Introducción: Los talares son pequeños bosques xerofíticos dominados por los árboles *Celtis tala* y *Scutia buxifolia*, característicos de las pampas en el noreste de la Provincia de Buenos Aires, Argentina. Los talares coexisten con áreas que enfrentan presiones urbanas y agrícolas significativas, lo que ha llevado a su empobrecimiento, fragmentación e incluso extinción local.
Objetivo: Caracterizar un talar resiliente ubicado en un bosque suburbano invadido por vegetación exótica, para identificar los patrones y procesos que actualmente afectan a las especies leñosas nativas.
Métodos: El bosque (55 ha) se dividió en siete sitios de muestreo. Hemos evaluado la abundancia y las variables de la fisiognomía de las especies leñosas nativas mediante búsquedas aleatorias (enfoque 1). Para las especies leñosas exóticas, utilizamos una combinación de transectos y cuadrículas dentro del bosque y en calles aledañas a él (enfoque 2). Los análisis incluyeron estadística descriptiva, cálculo de índices, y comparaciones mediante establecimiento de límites de confianza y pruebas Chi-cuadrado con corrección de Yates.
Resultados: En total, el tiempo dedicado fue de 118.5 h por miembro del equipo para el enfoque 1 y 48 h por miembro del equipo para el enfoque 2. Dos especies nativas dominan el parche talar estudiado: *C. tala* (88 %) y *S. buxifolia* (11 %), mientras que otras especies leñosas nativas fueron extremadamente raras (1 %). Las especies leñosas exóticas dominaron estructural y numéricamente. Observamos variación en la altura de *C. tala* y *S. buxifolia* entre los rodales forestales, siendo más comunes de menos de 3 m. El Índice de Manto indicó competencia lumínica por la sombra causada por especies exóticas grandes y de rápido crecimiento. Hubo daño debido a la caída de ramas de eucalipto solo en *C. tala*, aunque en un porcentaje bajo. En cuanto a la presencia de un hábito arbustivo, *S. buxifolia* tuvo el 20 % y *C. tala* 15 % de individuos con forma arbustiva, siendo algunos casos variables entre los rodales forestales. Proponemos varias estrategias de manejo tendientes a favorecer a las especies nativas sobre las exóticas.
Conclusión: El bosque estudiado es un ecosistema híbrido con rangos de neo-ecosistema que implica una serie de problemas de conservación para los remanentes de leñosas nativas allí presentes. Es altamente valioso para la conservación de los talares, particularmente para una de sus especies más emblemáticas, *S. buxifolia*, la cual está prácticamente ausente fuera del área de bosque.

Palabras clave: manejo adaptativo; resiliencia de bosques; ecosistemas híbridos; bosques suburbanos; pérdida de hábitat; leñosas nativas.

INTRODUCTION

Grasslands represent the climax community of the Pampa ecoregion, which spans Brazil, Uruguay, and Argentina. They consist of vast pastures occasionally interrupted by marshes, meandering streams, shallow lagoons, and small xeric forest patches that thrive in well-drained soils. In the Southern Pampa, these patches, locally referred to as talar forest patches (TFPs), are found in three sub-ecoregions: Rolling Pampa, Low-land Pampa, and Hill-land Pampa (Cabrera, 1971; Matteucci, 2012; Parodi, 1940). TFPs are dominated by *Celtis tala* Gillies ex Planch. and *Scutia buxifolia* Reissek, two species with a relatively wide distribution in Southern Bolivia, Paraguay, Northern Argentina, Southern Brazil, and Uruguay. However, dense populations of these species are primarily restricted to the Southern range of Argentina, particularly Buenos Aires province. Within TFPs, *C. tala* and *S. buxifolia* often co-dominate and are accompanied by less abundant species (Cabrera, 1971). Due to the relatively recent formation of Pampas soils after the last significant sea level drop (late Holocene, approximately 3000 years ago: Cavalotto, 2002), TFPs can be considered young forest communities. A recent study found that TFP diversity is higher in Northeastern patches due to their proximity to Northern forest ecoregions and hygrophilous forests, with diversity decreasing toward the South and West (Torres-Robles, 2009).

Pre-Hispanic settlers utilized TFPs in various ways (Paleo et al., 2002). However, during the Hispanic period entire trees were used for firewood, construction, and fencing; fruits were harvested for poultry feed; and cattle intrusion facing an increasing overexploitation that led to general impoverishment of the community (Pochettino et al., 2014). The decline in TFP species and the loss of many patches were noted early by pioneer researchers (Bruch, 1937; Parodi, 1940). Today, few TFPs are effectively protected, and most are either impoverished or extinct due to various threats (Fundación de Historia Natural Félix de Azara, 2006a; Guerrero, 2019; Morello, 2006). Over the past 30 years, conservationists have focused on the problems affecting TFPs, leading to new research and proposals for effective protection measures (Arturi, 1997; Arturi et al., 2006; Franco et al., 2018). These conservation and research efforts have highlighted two key issues: (i) invasion by exotic plants and transformation of TFPs into hybrid ecosystems, and (ii) generation of young TFPs that established themselves (colonizer TFPs) in areas where the soil has been disturbed or where exotic trees were introduced as part of the Pampas' lignification process (Apodaca & Guerrero, 2019; Morello, 2006). Globally, implanted forests that support subsistence or facilitate the creation of new native patches are a widespread phenomenon (Lugo, 1997; Parrotta, 1992). Radical ecosystem transformations have significantly altered native ecosystems worldwide. Such transformations have sparked theoretical debates and diverse conservation practices in these increasingly degraded ecosystems. The concept of a neo-ecosystem (or transformed ecosystem: Aronson et al., 2014; or emerging ecosystem: Clewell & Aronson, 2013), first proposed by Chapin & Starfield (1997), refers to a system of biotic, abiotic, and social factors that, due to human influence, differs from the historically dominant one. Neo-ecosystems tend to self-organize and develop new characteristics without intensive human management (Hobbs et al., 2013). The challenge lies in determining where each system falls on the gradient between an original state and a neo-ecosystem. Miller & Bestelmeyer (2016) stress the importance of identifying irreversibility thresholds in hybrid ecosystems to guide ecological restoration efforts. They also argue that, even if restoration is not feasible, controlling exotic species-whether for eradication or not-should be a key consideration. Managing exotic species, especially the most harmful ones, can reduce pressure on natives and potentially provide valuable functions if kept within acceptable abundance levels (Belnap et al., 2012; Davis et al., 2011; Simberloff, 2011). The aims of this study were: 1) to assess the composition, abundance, and physiognomy of a TFP within a suburban forest extensively invaded by exotic plant species, and 2) to estimate the abundance of these woody exotics in the TFP and its surroundings. We believe that identifying patterns in the physiognomy, abundance, and local distribution of native woody plants will help us understand the impact of invasive species and guide effective remediation and adaptive management strategies.

MATERIALS AND METHODS

Study Site: Las Banderitas Forest spans 55 hectares of forest and 6 hectares of shrubs and grasses. Located Northwest of La Plata city, near the headwaters of Carnaval stream (34°54'58" S & 58°06'10" W), it is an ecosystem where native and exotic woody plants coexist (Fig. 1). The site lies in a suburban area with three dispersed population centers covering 1 200 hectares of heavily modified Rolling Pampa landscapes. The dominant land uses are intensive, including 70 hectares of horticulture, 254 hectares of greenhouse floriculture, 35 hectares of soybean and corn fields, and smaller areas for beekeeping, poultry farming, small cattle and sheep farms, and pig breeding. The remaining area consists of abandoned greenhouses and non-productive soil, which has been colonized by native and exotic vegetation, sometimes forming hybrid grasslands, shrub lands, or small patches of exotic woody forest. Urbanization is minimal, lower than 1 house / hectare, and most pathways are unpaved. The area is undergoing continuous change due to city expansion, which is encroaching on rural and natural spaces, creating stressful social and

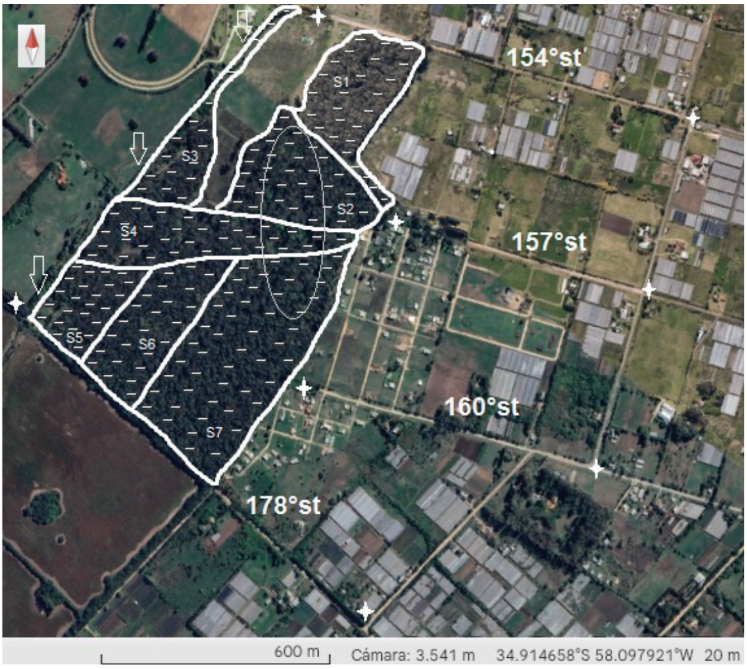

Fig. 1. Study site. Note the seven stands (S1-S7) into which the 55 ha of forest were divided and the converging streets (154, 157, 160 and 178). White stars at the beginning and end of each street indicate the area in which woody plants were sampled on the street shoulders out of the forest. Arrows indicate the course of the Carnaval stream. The area encircled represents the influence area of the central wetland within the forest. The white lines represent the approximate position of transects within stands. North is indicated by the red cone in the upper left corner of the figure.

ecological sceneries (Frediani, 2013; Matteucci & Morello, 2009).

Sampling design: We divided the 55 hectares of forest into seven smaller stands (S1: 8 ha, S2: 10ha, S3: 4 ha, S4: 8.5 ha, S5: 6 ha, S6: 6.5 ha, S7: 12 ha), using existing paths and open forest areas for demarcation (Fig. 1). Data collection involved two approaches: (1) measuring the abundance and physiognomic variables of native trees through random searches inside each stand, and (2) using transects and grids to sample exotic and native woody plants for assessing species abundance both inside and outside the forest. In both cases the search effort was measured in hours per person (we always used two people), with each team member given hours based on the search area (the larger the stand area, the more hours were dedicated proportionally). In total the complete work related to each approach involved 118.5 hours / team member in approach 1and 48 hours /team member in approach 2. We excluded shrubs from the genera *Pavonia* and *Baccharis* from sampling and analysis due to their extensive cover and abundant basal branching, which made individual identification difficult and required different methodologies. These genera will be included in future studies where understory vegetation will be addressed using appropriate methodologies and analyses.

Approach 1: Involved three phases, each covering the entire forest area. Phase 1 (initial count) included 15 visits between 22 August 2020, and 5 November 2020. Phase 2 (first re-count and new detections) comprised 56 visits from 13 November 2020 to 22 April 2022. Phase 3 (second re-count and new detections) consisted of eight visits between 30 June 2022 and 8 August 2022. We marked all sampled native woody species with a paint dot on the main trunk to prevent data duplication and recorded them based on (a) growing habit (arborescent or bushy), (b) height (meters) from ground level to the top edge of the crown, (c) crown diameter (meters), (d) trunk perimeter (centimeters) at a point between 0.5 and 1 meter in height (measured on the dominant trunk for bushy individuals), and (e) mode of growth (erect or comb-shaped). We also recorded standing dead individuals caused by lightning, excess water, or unknown reasons. We measured length variables using a semi-rigid 10 m long tape. The sampling method used in this approach was random. We ceased random sampling when we accounted for nearly all *C. tala* and *S. buxifolia* individuals within each stand. In other words, while the sampling method was random, the outcome approached a census. We used the ratio of recounted individuals to new detections to estimate the completeness of sampling within each stand.

Approach 2: Involved sampling using transects and grids. Within the forest, transects (20 x 4 meters; oriented Southeast-Northwest) and grids (placed at the beginning, middle, and end of each transect) were uniformly positioned every 50 meters within each stand to ensure consistent coverage across all stands, covering 25 % of each stand's area. Outside the forest, transects were placed on both sides of the four roads leading into the forest, ensuring almost complete coverage of the entire road area with minimal gaps (Fig. 1). Transects measured the abundance and height class frequency of woody species > 1 m tall. We measured the abundance of woody species < 1 m tall within 2 m^2 circle grids established at beginning, middle, and end of each transect. Along the transects on the streets out of the forest, we assessed plant abundance but not the height class frequency of all woody plants since shoulders constantly suffer from pruning by municipality operators.

Data Analysis: We used data on native woody species inside the forest to describe the physiognomy and abundance of dominant native trees. The dataset included three variables: height, crown diameter, and trunk perimeter. Heights were categorized into six classes for analysis: 1 (< 1 m), 2 (1-2.99 m), 3 (3-4.99 m), 4 (5-6.99 m), 5 (7-8.99 m), and 6 (> 9 m). The height of *C. tala* and *S. buxifolia* was statistically described using measures of central tendency (arithmetic mean) and dispersion (standard deviation) and compared between forest sites by establishing confidence limits for the mean of each species per site. The comparisons of the means were evaluated using the confidence intervals of the means with an alpha = 0.05 (the level used for the graphical part) and also at an alpha = 0.01 (to have greater confidence about the detected differences). We calculated the Mantle Index (MI = Crown Diameter / Height, Burger, 1939) to assess how tree crowns relate to lighting, with values near zero indicating asymmetrical crowns. To describe (using descriptive statistics and graphs) and analyze (by calculating confidence intervals for the mean) the index we followed the same procedure in the dominant natives, *C. tala* and *S. buxifolia*. For this purpose, we excluded trees shorter than 3 meters tall. Trunk diameter was derived from trunk perimeter and used to estimate the basal area of trees in height classes 2 to 6 using the formula Basal area = $(\pi / 4) * \text{trunk diameter}^2$. The basal area served as a complementary measure of tree density.

Chi-square tests with Yates' correction were used to evaluate differences in the percentages and proportions of the most abundant native woody plants, specifically in relation with standing dead trees, trees growing horizontally due to crushing by eucalyptus branches, and specimens with a shrubby growth form.

The abundance of native and exotic woody species inside the forest was addressed based on the data obtained from transect sampling. We assessed abundance by calculation of the Relative Importance Index (RII: Pinkas et al., 1971), an index of common usage in studies of trophic ecology, here adapted to evaluate the importance of each species. The index was calculated for each species using the variables

% FO (Percentage Frequency of Occurrence), % FH (Percentage Frequency of Height), and % FN (Percentage Frequency of Numerosity). The % FO variable represents the percentage of transects into which a particular species was found relative to the total number of transects and grids conducted. The % FH variable is the sum of the products between the mean of each height class and the abundance of each species within each height class, expressed globally as a percentage of the total height value for each species. This was the only variable modified from the original RII, which uses volume instead of height. Finally, the % FN variable is the abundance of each species expressed as a percentage of the total abundance of all species. The RII values of each species were ranked as a percentage relative to the highest RII value to better visualize the importance of each species and facilitate comparisons. Grid sampling was employed to assess the abundance of saplings within the forest.

We used Statistica 8.0 (StatSoft, 2007) and SigmaPlot 10.0 (Systat Software, 2006) for data analysis and graphing, with significance set at $p < 0.05$. The entire dataset is available at Alcalde (2024). Once you have accessed the page, you should enter the full name of the first author into the search bar. This will then display a set of research data from which you can select the desired one.

RESULTS

We detected a total of 4 029 native woody trees and bushes across the seven forest stands: 3 553 *C. tala*, 454 *S. buxifolia* (present in all stands), 30 *Abutilon grandifolium* (Will.) Sweet (S5), 11 *Sambucus australis* Cham. & Schltdl. (bushy and < 5 m tall, in S1-S3, S5, and S7), four *Senegalia bonariensis* (Gillies ex Hook. & Arn.) Seigler & Ebinger (S1, S2, and S4), three *Erythrina crista-galli* L. (S2), and two *Parkinsonia aculeata* L. (S5). Since *C. tala* and *S. buxifolia* accounted for 99 % of the woody natives, we focused our analysis on these dominant species.

C. tala had a higher density than *S. buxifolia* across all forest stands (Table 1). The late species was so scarce at stands 3 to 7 that these stands had to be grouped to obtain satisfactory degrees of freedom for all comparisons and analyses conducted. *S. buxifolia* exhibited higher density in S1 and S2, while *C. tala* was denser in S3, followed by S5 and S1. The basal area of *C. tala* for the entire forest was 860.27 m^2 (15.6 m^2/ha), with the lowest value in S6 (10.61 m^2, 1.6 m^2/ha) and the highest in S3 (351.76 m^2, 87.9 m^2/ha) (Table 1).

C. tala consistently dominated numerically over *S. buxifolia* across all stands, with particularly dominance in S3 to S7 where *S. buxifolia* was extremely infrequent (Table 1). Analysis of the height for *S. buxifolia* and *C. tala* revealed that trees were less than 3 meters height were predominant (Fig. 2, Table 1). The overall height pattern of both species was similar, with both dominated by individuals shorter than three meters in height. The forest stands that replicated the global pattern of both species, and partly explained it, were the stands at the forest front (S1, S2 for both species, S3 for *C. tala*), with a greater number of taller trees observed at the stands on the opposite end of the forest (Fig. 2, Table 1).

The mean height of *S. buxifolia* was not different between S1 and S2, but both differed from the mean of the groupings of the stands at the opposite end of the forest (S3-7) using a 99 % confidence level (Fig. 3A).

When comparing the mean heights of *C. tala* using stringent confidence intervals (99 %), we also detected general height differences, with lower means at the NE-NW front sites of the forest (S1, S2, S3, and S5, being S2 the place with the higher mean of them) compared to the higher mean heights at the SE-SW sites on the opposite end (Fig. 3B).

The comparison through the establishment of 99 % confidence limits of the mean mantle index did not detect differences between forest stands for *S. buxifolia* equal to or greater than three meters in height (the differences were not detected even when using a more relaxed confidence level of 95 %, Table 2 and Fig. 4A). However, it can be said that the mean mantle index trends from higher values at the forest front in

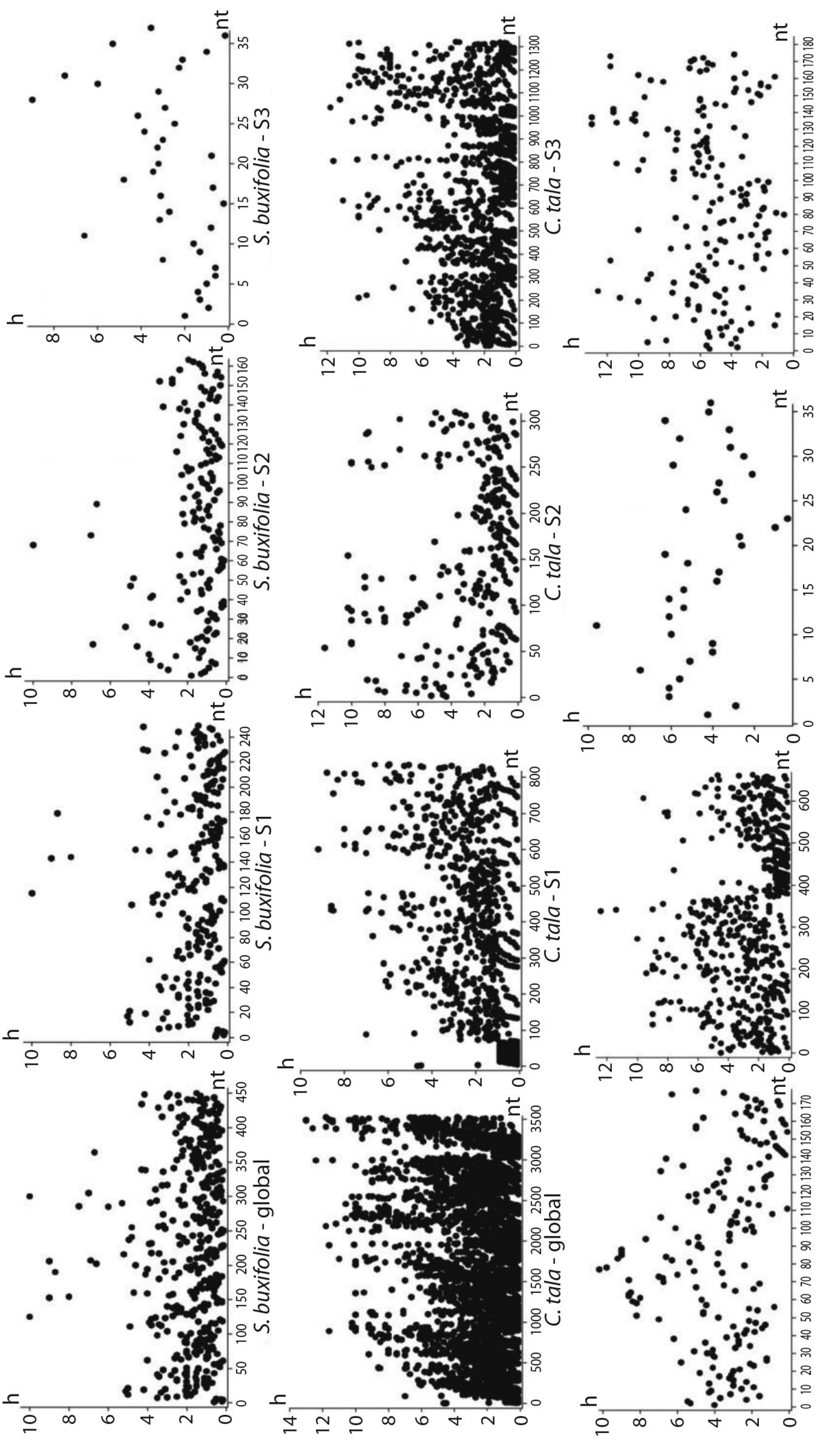

Fig. 2. Number of *S. buxifolia* and *C. tala* trees (X-axis) of different height (Y-axis) for the entire forest and stand to stand (*S.buxifolia* for S3 to S7 were presented together, see text).

Table 1
Global and stand to stand values of tree density (td), basal area (ba), and height of *C. tala* and *S. buxifolia* from the studied forest.

stand	n	td	ba	mean H (min-max)	- 95	+ 95	sd
S. Buxifolia							
global	454	8.2	8.3 (0.15/ha)	1.6 (0.12-10)	1.5	1.8	1.5
S1	252	31.5	4.5 (0.56/ha)	1.6 (0.15-10)	1.4	1.8	1.4
S2	162	16.2	2.4 (0.24/ha)	1.5 (0.12-10)	1.2	1.7	1.4
S3-7	40	1	1.27 (0.03/ha)	2.7 (0.15-9)	2.08	3.3	2
C. Tala							
global	3 553	64.1	860.2 (15.6/ha)	2.7 (0.1-13)	2.6	2.8	2.3
S1	839	104.5	141.4 (17.6/ha)	2.3 (0.1-9.2)	2.2	2.4	1.7
S2	311	31	116.4 (11.6/ha)	2.7 (0.1-11.6)	2.5	3.08	2.5
S3	1 322	330.5	351.7 (87.9/ha)	2.5 (0.1-11.8)	2.3	2.6	2.2
S4	183	20.8	36.6 (4.3/ha)	3.5 (0.1-10.2)	3.2	3.9	2.2
S5	667	112.3	111.4 (18.5/ha)	2.5 (0.1-12.4)	2.3	2.7	2.1
S6	37	5.53	10.61(1.6/ha)	4.5 (0.4-9.6)	3.9	5.1	1.8
S7	184	14.5	91.87 (7.6/ha)	5.5 (0.5-13)	5.1	5.9	2.8

td: trees/ha, ba: crude and referred to 1 ha, n: number of trees, min-max: minimum height – maximum height, - 95 % and + 95 %: confidence limits of mean, sd: standard deviation.

Table 2
Values and descriptive statistics of the mantle index for *S. buxifolia* and *C. tala* specimens equal to or greater than 3 meters in height, presented for the entire forest and broken down by stand (S).

stand	n	mean (min-max)	- 95	+ 95	sd
Scutia buxifolia					
global	71	0.76 (0.23-1.38)	0.7	0.82	0.24
S1	36	0.8 (0.23-1.38)	0.71	0.89	0.27
S2	17	0.67 (0.31-1.15)	0.55	0.79	0.23
S3-7	18	0.77 (0.46-1.05)	0.68	0.86	0.17
Celtis tala					
global	1 285	0.66 (0.04-1.88)	0.62	0.67	0.23
S1	246	0.65 (0.2-1.47)	0.62	0.68	0.22
S2	67	0.69 (0.2-1.54)	0.62	0.75	0.26
S3	431	0.66 (0.2-1.78)	0.64	0.68	0.24
S4	100	0.71 (0.3-1.4)	0.66	0.76	0.24
S5	237	0.65 (0.04-1.8)	0.62	0.68	0.23
S6	30	0.71 (0.4-1.51)	0.63	0.8	0.23
S7	143	0.61 (0.29-1.19)	0.57	0.64	0.18

n: number of trees, (min-max): (minimum-maximum), -95 and + 95: confidence limits of the mean, sd: standard deviation. Note that for the first species, neighboring stands S3 to S7 were grouped due to the low number of specimens in each one.

S1 and forest bottom in S7 (indicating crown imbalance and higher competition for light) toward higher values at the center of the forest (and vice versa). The mean mantle index of *C. tala* varied across forest stands, but significant differences, as determined by 99 % stringent confidence limits, were only noted between S4 and S7 (Table 2, Fig. 4B).

Of the two most abundant native woody species, the phenomenon of standing dead trees was observed only for *C. tala* at stands on the NE-NW edge of the forest (S1, S2) and at the

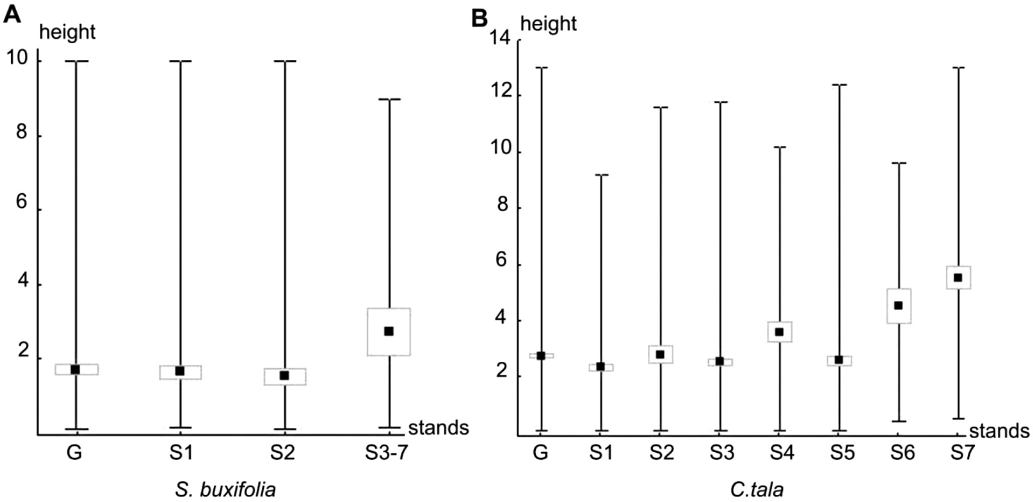

Fig. 3. Box plots of the height. **A.** Of *S. buxifolia*. **B.** Of *C. tala.* Globally for the forest (G) and from each forest stand (S). Black squares: mean, grey boxes: 95 % confidence limits of mean, extreme ends of vertical lines: minimum and maximum height values.

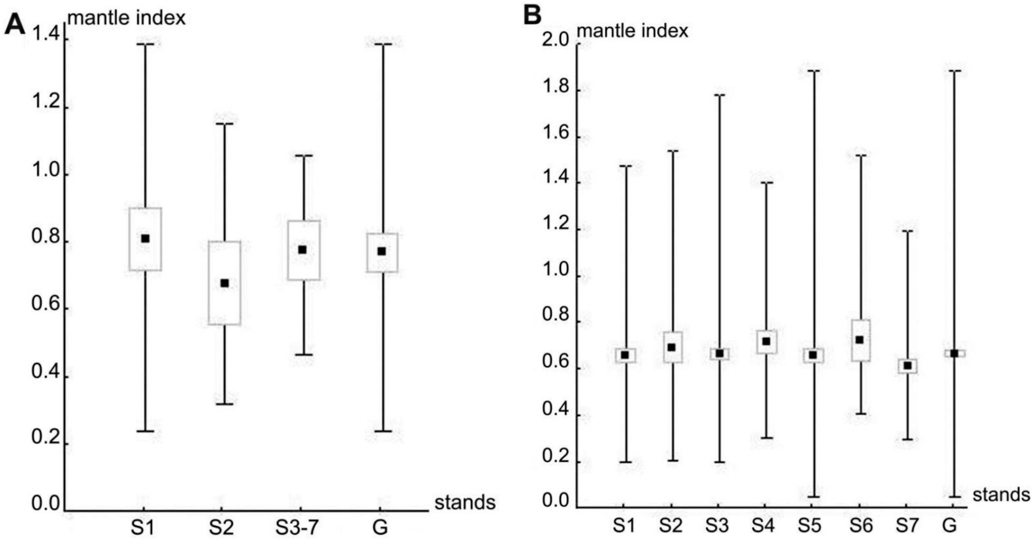

Fig. 4. Box plots of the Mantle Index values. **A.** Of *S. buxifolia*. **B.** Of *C. tala*. Corresponding to tress equal or higher than three meters in height. References as in Fig. 3.

stands on the opposite end (S5, S6, and S7), but was not observed at the central forest stands. The percentages of standing dead *C. tala* were low across stands (S1: 0.4 %, S2: 1.9 %, S5: 5 %, S6: 2 %, S7: 3 %), and comparisons using the Chi-square test with Yates' correction did not detect differences between them. Another phenomenon we detected in *C. tala* but not in S. *buxifolia* was the presence of trees growing flattened with their main axis parallel or bent relative to the ground due to the falling of large branches, primarily from *E. camaldulensis*. The percentage of flattened *C. tala* trees in each forest stand was low (S1: 2 %, S2: 6 %, S3: 1.5 %, S4: 2 %, S5: 4 %, S6: 5 %, S7: 2 %), and the comparison of these values using the Chi-square test with Yates' correction did not detect significant differences between any pair.

Overall, *S. buxifolia* exhibited 20 % of individuals with a shrub-like form, with similar values across each forest stand (S1: 17 %, S2: 25 %, S3-7: 10 %). The comparison using the Chi-square test with Yates' correction detected differences only between S2 and S3-7 ($p < 0.05$). Considering all sites, *C. tala* showed 15 % of individuals with a shrub-like habit. Comparisons between stands using the Chi-square test with Yates' correction detected differences between stands at the SE-SW end of the forest (except for S6 which had the lowest percentage) and stands at the opposite end (NE-NW forest edge) (Table 3).

The total number of woody species detected inside and outside the forest was similar (39 inside with seven native and 40 outside with eight native), but the species composition differed (Table 4). Streets 154 and 160 each had 21 species, while Street 160 had the fewest (19) and Street 178 had the most (29). Exotic species were dominant, especially *C. laevigata*, *L. lucidum*, *C. australis*, *M. alba*, *L. sinense*, *G. triacanthos*, and *P. alba*. The former was the most prevalent and, together with *L. lucidum*, both dominated on the four streets. *Gleditsia triacanthos* was dominant only on Street 178 (limiting stands 5 to 7, Fig. 1). *Fraxinus pennsylvanica* was dominant only on Street 157 (limiting stands 1 and 2, Fig. 1). Of the natives, *C. tala* was present on all streets, but other native species were mostly limited to Street 178, with one species (*V. caven*) only found on Street 160 (limiting stands 4 and 7, Fig. 1).

Both full RII_f and its simplified RII_s (without using height frequency) version coincided in provided the same global ranking of the species inside the forest (Table 4). When comparing stand by stand, some of them showed changes in the ranking of the top five species in each case (Fig. 6). These differences occurred at S1 to S4, where some large species (generally *E. camaldulensis*) moved to less important positions and were replaced by species with greater numerical abundance but not as tall (Table 4 and Fig. 5 for discordances between RII_f and RII_s values).

The simplified % RII_s identified the following five dominant species (listed in descending order according to RII_s value): *L. sinense*, *C. laevigata*, *E. camaldulensis*, *M. alba*, and *L. lucidum* (inside forest), and *C. laevigata*, *L. lucidum*, *M. alba*, *L. sinense*, and *G. triacanthos* (outside forest). The exclusion of *E. camaldulensis* from the top five species outside the forest and its replacement by *G. triacanthos*, which turned out to be particularly abundant in one of the adjacent streets, explains the only differences detected by the RII_s between inside and outside the forest. Finally, grid sampling within the forest detected an almost absolute sapling abundance of *L. sinense* (86 %), followed by *L. nobilis*

Table 3

Percentage of *C. tala* with shrub-like habit we found at each forest site (diagonal line) and its comparison between stands using Chi-square test with Yates' correction.

	S1	S2	S3	S4	S5	S6	S7
S1	11 %						
S2	NS	9 %					
S3	NS	NS	9 %				
S4	NS	*	*	22 %			
S5	*	**	**	NS	26 %		
S6	NS	NS	NS	**	**	5 %	
S7	*	**	**	NS	NS	**	26 %

One asterisk represents significant differences ($p < 0.05$), two asterisks indicate highly significant differences ($p < 0.01$), and NS represents no significance in the comparison.

Table 4

Percentage values of frequency of occurrence (% FO), numerical frequency (% FN), height frequency (% FH), % RII full (% RII$_f$) and RII simplified (% RII$_s$) of the woody species detected trough transect sampling.

Species	% FO	% FN	% FH	% RII$_f$	% RII$_s$
Global forest					
Oleaceae: *Ligustrum sinense* Lour.	78.6	42.4	20.5	100	100
Cannabaceae: *Celtis laevigata* Willd.	90.7	28.7	23.4	96	78.1
Myrtaceae: *Eucalyptus camaldulensis* Dehnh.	86.4	8.01	35.5	76	20.6
Moraceae: *Morus alba* L.	60.1	5.5	7.5	16	9.9
Oleaceae: *Ligustrum lucidum* W. T. Aiton	48.05	3.4	2.1	5	4.9
Cannabaceae: *Celtis tala**	31.06	2.8	2.1	3	2.6
Cannabaceae: *Celtis australis* L.	29.1	1.2	1.3	2	1.07
Sapindaceae: *Acer negundo* L.	14.5	2.4	2.8	2	1.07
Lauraceae: *Laurus nobilis* L.	22.3	1.7	1.1	1	1.01
Fabaceae: *Gleditsia triacanthos* L.	13.1	0.6	0.7	< 1	< 1
Lauraceae: *Cinnamomum glanduliferum (Wall.) Meisn.*	12.1	0.2	0.4	< 1	< 1
Verbenaceae: *Lantana camara* L.	8.2	0.6	0.2	< 1	< 1
Rhamnaceae: *Scutia buxifolia**	8.2	0.3	0.2	< 1	< 1
Rosaceae: *Pyracantha coccinea* M. Roem.	6.3	0.2	0.1	< 1	< 1
Fabaceae: *Racosperma melanoxylon* R. Br.	3.3	0.2	0.4	< 1	< 1
Salicaceae: *Populus alba* L.	1.45	0.45	0.36	< 1	< 1
Salicaceae: *Salix babylonica* L.	0.9	0.07	0.2	< 1	< 1
Pittosporaceae: *Pittosporum tobira* (Thunb.) W. T. Aiton	1.9	0.04	0.02	< 1	< 1
Arecaceae: *Phoenix canariensis* H. Wildpret	1.4	0.03	0.02	< 1	< 1
Arecaceae: *Syagrus romanzoffiana* (Cham.) Glassman	1.4	0.03	0.01	< 1	< 1
Fagaceae: *Quercus robur* L.	0.9	0.04	0.04	< 1	< 1
Adoxaceae: *Sambucus australis**	1.4	0.04	0.01	< 1	< 1
Oleaceae: *Fraxinus pennsylvanica* Marshall	0.4	0.07	0.05	< 1	< 1
Rosaceae: *Prunus serrulata* Lindl.	0.9	0.02	0.02	< 1	< 1
Casuarinaceae: *Casuarina cunninghamiana* Miq.	0.4	0.05	0.02	< 1	< 1
Anacardiaceae: *Schinus areira* L.	0.9	0.02	0.01	< 1	< 1
Arecaceae: *Livinstona australis* (R. Br.) Mar.	0.9	0.02	0.09	< 1	< 1
Cupressaceae: *Cupressus lusitanica* Mill.	0.9	0.02	0.009	< 1	< 1
Arecaceae: *Butia yatay* (Mart.) Becc.	0.4	0.01	0.004	< 1	< 1
Fabaceae: *Parkinsonia aculeata** L.	0.4	0.01	0.004	< 1	< 1
S1 (top five)					
C. laevigata	100	37	36	100	100
E. camaldulensis	88	11	27	46	26.7
L. sinense	73	22	11	32	42.5
M. alba	81	11	10	23	24.5
L. lucidum	54	6	4	8	9.1
S2 (top five)					
C. laevigata	97	25	22	100	80.03
L. sinense	72	41	18	95	100
E. camaldulensis	97	7	32	84	22.8
A. negundo	56	13	13	32	24.5
M. alba	75	6	9	25	14.8

Species	% FO	% FN	% FH	% RII$_f$	% RII$_s$
S3 (top five)					
C. laevigata	75	27	58	100	100
*C. tala**	69	25	9	36	85.8
M. alba	88	13	8	29	58.9
E. camaldulensis	31	11	16	13	16.9
L. lucidum	50	7	2	7	17.6
S4 (top five)					
L. sinense	91	72	40	100	100
E. camaldulensis	88	7	29	31	9.3
C. laevigata	68	11	15	17	11.1
M. alba	59	4	10	8	3.4
L. lucidum	50	2	2	2	1.6
S5 (top five)					
C. laevigata	100	46	27	100	100
E. camaldulensis	79	11	45	61	23.2
L. sinense	63	17	8	22	19.7
*C. tala**	50	7	5	8	7.2
M. alba	42	6	6	7	3.05
S6 (top five)					
L. sinense	88	55	23	100	100
E. camaldulensis	92	14	55	91	26.03
C. laevigata	96	19	12	44	38.1
L. lucidum	50	3	2	4	3.2
M. alba	27	2	3	2	1.2
S7 (top five)					
L. sinense	92	48	24	100	100
C. laevigata	96	33	26	87	72
E. camaldulensis	96	7	37	65	16.3
M. alba	58	3	5	7	3.2
L. lucidum	54	2	1	3	2.6
Street shoulders					
C. laevigata	100	40.7	-	-	100
L. lucidum	75	19.09	-	-	35.1
M. alba	100	7.4	-	-	18.8
L. sinense	100	1.1	-	-	14.4
G. triacanthos	100	4.4	-	-	11
*C. tala**	100	2.9	-	-	7.3
F. pennsylvanica	100	2.5	-	-	6.3
C. australis	50	4.5	-	-	5.5
P. alba	50	4.03	-	-	4.9
L. nobilis	100	1.1	-	-	2.7
R. pseudoacacia	75	1.01	-	-	1.8
C. cunningamiana	75	1.005	-	-	1.6
C. lussitanica	75	05	-	-	1
*S. bonareinsis**	25	1.5	-	-	< 1
E. camaldulensis	75	0.3	-	-	< 1
B. papyrifera	50	0.5	-	-	< 1

Species	% FO	% FN	% FH	% RII$_f$	% RII$_s$
Salicaceae: *Populus deltoides* W. Bartram ex Marshall	100	0.2	-	-	< 1
Meliaceae: *Melia azedarach* L.	100	0.1	-	-	< 1
A. negundo	50	0.2	-	-	< 1
R. melanoxylon	50	0.1	-	-	< 1
S. areira	75	0.1	-	-	< 1
Q. robur	50	0.1	-	-	< 1
B. forficata	50	0.1	-	-	< 1
Cupressaceae: *Cupressus sempervirens* L.	25	0.1	-	-	< 1
S. babylonica	75	0.03	-	-	< 1
P. coccinea	50	0.04	-	-	< 1
*S. australis**	25	0.07	-	-	< 1
E. japonica	50	0.03	-	-	< 1
*P. aculeata**	25	0.04	-	-	< 1
*V. caven**	25	0.02	-	-	< 1
P. granatum	25	0.02	-	-	< 1
L. camara	50	0.01	-	-	< 1
A. grandifolium	25	0.03	-	-	< 1
*S. buxifolia**	25	0.01	-	-	< 1
C. glanduliferum	25	0.01	-	-	< 1
Phytolaccaceae: *Phytolacca dioica** L.	25	0.01	-	-	< 1
Myrtaceae: *Callistemon citrinus* (Curtis) Skeels	25	0.01	-	-	< 1
Moraceae: *Ficus cairica* L.	25	0.01	-	-	< 1
Simaroubaceae: *Ailanthus altissima* (Mill.) Swingle	25	0.01	-	-	< 1
Species within forests but not detected trough transects					
Scrophulariaceae: *Myosporum laetum* G. Forst.	Present in S2				
Rosaceae: *Raphiolepis umbellata* (Thunb.) Makino	Present in S3				
Grossulariaceae: *Ribes rubrum* L.	Present in S1, S5, and S6				
Fabaceae: *Senegalia bonaeriensis**	Present in S1-S4				
Moraceae: *Maclura pomifera* (Raf.) C. K. Schneid.	Present in S1, S2, and S3				
Fabaceae: *Robinia pseudoacacia* L.	Present in S6 and S7				
Fabaceae: *Erythrina crista-galli**	Present in S2				
Rutaceae: *Citrus trifoliata* L.	Present in S5				
Moraceae: *Broussonetia papyrifera* L. (Vent.)	Present in S5				
Malvaceae: *Abutilon grandifolium**	Present in S5				

% FH: not calculated for transects out of the forests: see explanation in M&M section, % RII$_f$: using the three variables, % RII$_s$: using % FO and % FN only. Note that the full % RII$_f$ was not calculated for street´s species: see explanation in M&M section. The ten species listed at the end of the table were not detected through transect samplings inside the forest due their extremely low abundance. In addition, some trees were recently planted in few quantities (< 10) in the forest borders (thus not listed in the table): *Gynkgo biloba* L. (Gynkgoaceae), *Lagestroemia indica* (L.) Pers. (Lythraceae), *Cydonia oblonga* Mill. (Rosaceae), *Eriobotrya japônica* (Thunb.) Lindl (Rosaceae), *Persea Americana* Mill. (Lauraceae), *Bauhinia forficata* Link (Fabaceae), *Firmiana simplex* (L.) W. Eight (Malvaceae), *Schinus longifolius** Lindl. (Speg.) (Anacardiaceae), *Ceiba sp*eciosa (A. St.Hill.) Ravenna (Malvaceae), *Enterolobium contortisiliquum* (Vell.) Morong (Fabaceae). Asterisks following species names denote the species is native of the area.

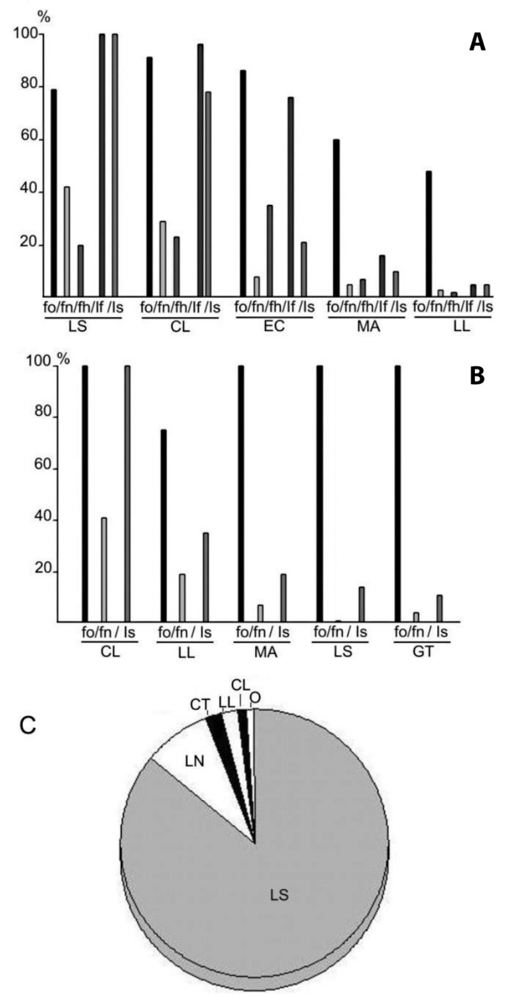

Fig. 5. Percentage values (Y-axis) of occurrence frequency (fo), numerical frequency (fn), height frequency (fh), relative importance index in its full version (If), and its simplified version (excluding height frequency, Is). **A.** For the five species identified as most important by this index in the transects within the forest. **B.** outside the forest. **C.** The pie chart shows the percentage of saplings detected through grid sampling within the forest. References: CL: *C. laevigata*, CT: *C. tala*, EC: *E. camaldulensis*, GT: *G. triacanthos*, LL: *L. lucidum*, LN: *L. nobilis*, LS: *L. sinense*, MA: *M. alba*, O: other species (*A. negundo*, *C. glanduliferum*, *E. camaldulensis*, *G. triacanthos*, *L. camara*, *M. alba,* palm indet., *P. alba*, *P. coccinea*, and *S. buxifolia*.

(8 %), the pair *C. tala*-*L. lucidum* (2 % each), and *C. laevigata* (1 %), while the remaining percentage (1 %) corresponded to species with very low representation of saplings (Fig. 5C).

The stand by stand discrimination of the species that presented saplings is as follows: (S1) *C. glanduliferum*, *L. sinense* (dominant), and *S. buxifolia*; (S2) *A. negundo*, *C. glanduliferum*, *L. sinense* (dominant), *M. alba*, palm indet., and *S. buxifolia*; (S3) *C. laevigata*, *C. tala* (dominant), *E. camaldulensis*, *L. camara*, *L. lucidum*, and *L. sinense* (dominant); (S4) *E. camaldulensis*, *L. sinense* (dominant), and *P. coccínea*; (S5) *G. triacanthos*, *L. camara*, *L. sinense* (dominant), and *P. alba;* (S6) *E. camaldulensis*, *L. camara*, and *L. sinense* (dominant); and (S7) *L. sinense*.

DISCUSSION

This study inventories and establishes a baseline for the species' composition and abundance of woody plants in a suburban forest under urban pressure and suggests management strategies to enhance native species. We characterized the forest as a neo-ecosystem based on Clewell & Aronson (2013) and Aronson et al. (2014) ecosystem concepts. These ecosystems, described by Chapin & Starfield (1997), arise from the interplay of biotic, abiotic, and social factors, creating a new ecosystem distinct from the historical one due to human influence. They tend to self-organize and sustain themselves with minimal human intervention (Hobbs et al., 2013).The studied forest exhibited the following characteristics: (1) *C. tala* was the dominant native woody species both inside and outside the forest, followed by *S. buxifolia* (which was nearly absent outside the forest), being other native species more abundant in the surroundings than in the forest; (2) there were 33 woody exotic species (Table 4), five of them being structurally and numerically dominant over native species; (3) most *S. buxifolia* and *C. tala* individuals were < 3m in height; and (4) the mantle index, which indicates imbalanced growth due to light competition, revealed intense competition for light in both species, particularly noticeable in young individuals. Characteristics 2-4, combined with the apparent greater success of exotic species in being dispersed by frugivorous birds (see disclaimers views in Traveset 2015;

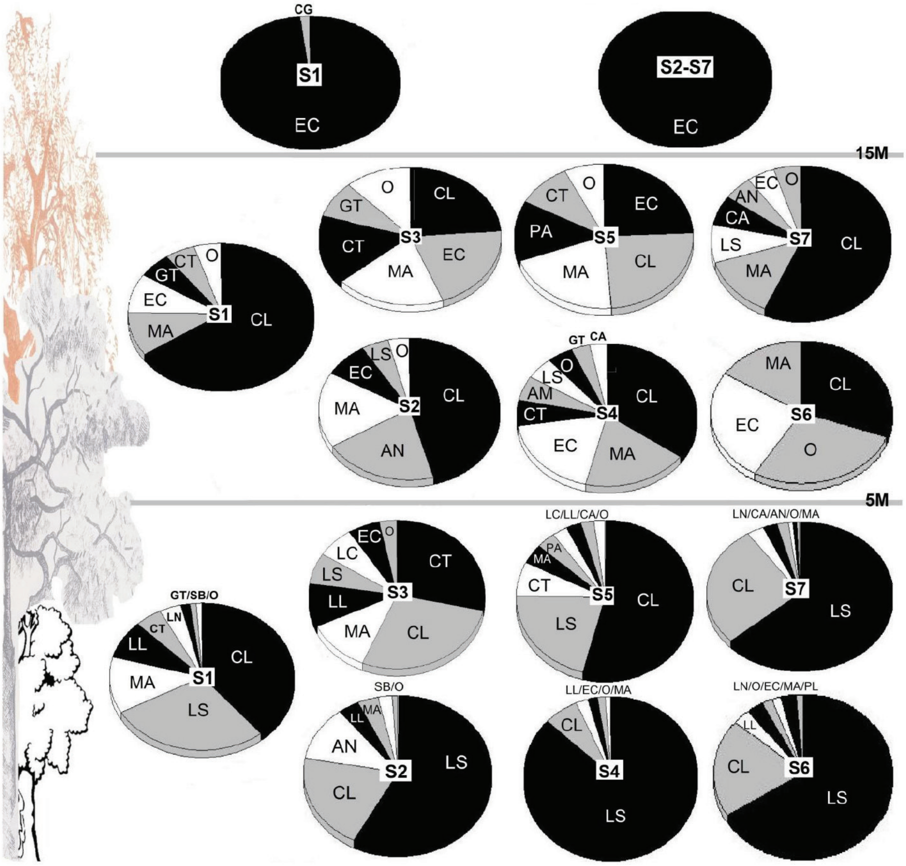

Fig. 6. Profile of the forest strata and their composition considering the woody species detected in the sampling by transects according to their numerical frequency and height. The pie charts at the base of the figure correspond to the lower shrub layer and the regeneration of trees from the lower strata (< 5m). The pie charts in the center of the figure represent the percentage composition of abundance of the middle layer, dominated by specimens of species whose height does not exceed 15m, but also with the presence of growing specimens from the next layer. Finally, the pie charts in the upper portion of the figure illustrate the composition of the upper layer with trees over 15m tall. References: AN: *A. negundo*, CA: *Celtis australis*, CG: *Cinnamomum glanduliferum*, CL: *Celtis laevigata*, CT: *Celtis tala*, EC: *Eucalyptus camaldulensis*, GT: *Gleditsia triacanthos*, LC: *Lantana camara*, LL: *Ligustrum lucidum*, LN: *Laurus nobilis*, LS: *Ligustrum sinense*, MA: *Morus alba*, O: species with less than 10 individuals (see discrimination above), PA: *Populus alba*, SB: *Scutia buxifolia*. Composition of less abundant species (OTHER) by stratum and stand: S1 < 5 m (CA-CG-EC-*Pittosporum tobira*-*Sambucus australis*-*Schinus areira*) / S1 / 5-15 m (AN-CA-CG-LL-LN-LS-*S. areira*-SB) / S2 < 5 m (*Butia yatay*-CA-CG-CT *Cupressus lusitanica*-EC-GT-LN-LS-*P. tobira*-*Pyracantha coccinea*) / S2 / 5-15 m (CA-CG-CT-GT-LL-LN-*Salix babylonica*-SB) / S3 < 5 m (GT-LN-PA-*P. coccinea*-*Racosperma melanoxylon*-*S. australis*) / S3 / 5-15 m (AN-CA-LL-PA-*R. melanoxylon*) / S4 < 5 m (AN-*Casuarina cunninghamiana*-CA-CT-CG-*Fraxinus pennsylvanica*-GT-LN-*Phoenix canariensis*-*P. tobira*-*P. coccinea*-*Quercus robur*-*R. melanoxylon*-SB) / S4 / 5-15 m (AN-CG-LL-PA-*Q. robur*-SB) / S5 < 5 m (CG-*C. lusitanica*-EC-GT-*P. aculeata*-*P. canariensis*-*Syagrus romanzoffiana*) / S5 / 5-15 m (CA-GT-LS-LL-*S. babylonica*) / S6 < 5m (CA-CT-CG-GT-*Livinstona australis*-LC-*Prunus serrulata*-*S. romanzoffiana*) / S6 / 5-15 m (CA-CG-GT-LL-LN-LS) / S7 < 5 m (CT-CG-EC-LC-*S. romanzoffiana*) / S7 / 5-15 m (CG-CT-LL-LN-SB).

Vergara-Tabares, et al. 2022) suggest that the studied forest have reached the threshold of irreversibility required to be considered as a neo-ecosystem, as proposed by Miller & Bestelmeyer (2016).

There were published many works evidencing dispersion of the plants by bird present in the study site (De la Peña, 2002; Guidetti, 2020; Murriello et al., 1993; Weyland et al., 2014). We observed this phenomenon during fieldwork Native birds (*Mimus saturninus*, *Pitangus sulphuratus*, *Turdus rufiventris*, *T. amaurochalinus*) have fed upon over exotic plants but also displayed a high consumption over both dominant natives, particularly over *S. buxifolia* during the coldest months. We observed two fructification periods in the late species (fall-winter and late spring) contrary to *C. tala* (late spring-summer).

We believe that the central wetland of the studied forest and the higher soils in the wetland surroundings may contribute to the heterogeneity within stands S1, S2, S4, and S7, surely favoring the survival of TFPs in these stands. Similarly, the elevated areas near the Carnaval stream's floodplain might explain the survival of TFPs in stands 3 and 5. The greatest abundance of *S. buxifolia* and the tallest trees of this species were found around the central wetland. Flooding in the wetland likely protected the ancient trees from logging and provided suitable conditions for the species. Ribichich (1996) reported that TFPs near the Rio de La Plata shoreline showed dominance of *S. buxifolia* over C. *tala*, while further from the coast, both species co-dominated, with *C. tala* becoming more dominant. This study demonstrated *S. buxifolia*'s tolerance to some flooding, suggesting that flooding limits *C. tala*. Indeed, TFPs closer to the river shoreline are nearer to the surface water table compared to more inland ones (Arturi, 1997). This pattern was also observed in *S. buxifolia* from Martin Garcia Island and Punta Lara Natural Reserve, where the species adapted to varying soil moisture levels, including some flooding (Arturi & Juarez, 1997).

The TFPs within the studied forest were significantly impacted by taller and faster-growing exotic species due to shading. We observed a significantly higher proportion of standing dead *C. tala* compared to none for *S. buxifolia*. This could be explained by two factors (which may even combine additively): (1) most exotic trees that provide significant shade are deciduous (like *C. tala*), allowing *S. buxifolia* (an evergreen species) to photosynthesize during the autumn and winter, and (2) *C. tala*'s intolerance to waterlogging versus *S. buxifolia*'s higher resistance to it (as mentioned in previous paragraphs), particularly in areas close to the central wetland. Highly invasive species often exhibit similar competitive behavior in their introduced environments across worldwide. Several exotic species we found in the studied forest were reported for various Argentine ecosystems (Fundación de Historia Natural Félix de Azara, 2006b; Zagel, 2006). *Ligustrum lucidum* was recorded invading the Punta Lara Natural Reserve' hygrophilous forests 79 years ago (Cabrera & Dawson, 1944), and by later years, the site was dominated by pure stands of *L. lucidum* (Dascanio & Ricci, 1988). This species has significantly impacted native woody species in coastal TFPs. Thirty years after invasion patches showed 80 % of individuals standing dead compared to the 20 % reported earlier (Franco et al., 2018; Goya et al., 1992). Ribichich & Protomastro (1998) suggested that the invasion of *L. lucidum* might have been accelerated by the exclusion of cattle in the 1980s, as cows previously grazed on this species.

We think that the abundance of each woody exotic in the studied forest is primarily influenced by their presence in areas outside the forest (a facilitating factor) and secondarily by subtle soil characteristics within the forest (a limiting factor). In that sense, the eradication of certain invasive woody exotics would be extremely difficult, considering the multiplicity of life history strategies (re-growth from stumps, root sprouting, abundant palatable fruits with highly germinal seeds, fast growth). Although there are no entirely successful examples of controlling invasive exotics once they are established (Aragón & Groom, 2003), some acceptable success has been achieved with significant operational and economic efforts. Examples include control of blackberries (Mazzolarri et al., 2011), privets and green ashes (Fontes, 2015), honey locusts (Sosa et al., 2015), and white mulberries (Torresin et al., 2013). The cost-benefit ratio of these efforts often makes them unfeasible in the long term, as even a slight relaxation can lead to re-colonization.

The Rolling Pampa ecosystems, especially around the densely populated Buenos Aires Metropolitan Area (Instituto Nacional de Estadística y Censos, 2012a; Instituto Nacional de Estadística y Censos, 2012b; Instituto Nacional de Estadística y Censos, 2016), have been heavily exploited and degraded over centuries, losing their original condition (Manuel-Navarrete et al., 2009; Matteucci & Morello, 2009; Morello & Mateucci, 2000). The region now consists of hybrid and neo-ecosystems with complex, not fully understood dynamics, yet they still support many valuable native species of high conservation value.

Miller & Bestelmeyer (2016) emphasized the importance of neo-ecosystems in successful conservation strategies. Effective management should focus on supporting desirable species rather than solely eradicating undesirable ones (Belnap et al., 2012; Davis et al., 2011; Seastedt et al., 2008; Simberloff, 2011). Given the challenges of restoring historical conditions, efforts should prioritize mitigating and containing invasive species while promoting local economic benefits through methods like pruning and native planting (Arturi et al., 2006; Doumecq & Arenas, 2018; Plaza-Behr et al., 2016; Vargas-Monter, 2012; Wilcox, 2000). Different pruning methods (selective, mixed, low) have been recommended to positively affect growth of native renewals, especially when applied to rotating small parcels and combined with native planting for environmental restoration (Administración de Parques Nacionales, 2007; Arturi et al., 2006; Franco et al., 2018; Plaza-Behr et al., 2016). Such practices would likely benefit the native species in the studied forest. Our study identifies viable populations of two declining species: the tree *S. buxifolia* and the Argentine flag butterfly *Morpho epistrophus argentinus* Fruhstorfer (Nymphalidae) recently reported there (Alcalde & Rodríguez-Allo, 2023), which depends on *S. buxifolia* as its host. The site also supports a ground orchid *Chloraea membranacea* Lindl. (Orchidaceae) and other threatened native herbs (Delucchi, 2006) like *Senecio selloi* (Spreng,) DC. (Asteraceae) and *Cypella herbertii* (Lindl.) Herb. (Iridaceae), providing hope for their conservation.

Ethical statement: the authors declare that they all agree with this publication and made significant contributions; that there is no conflict of interest of any kind; and that we followed all pertinent ethical and legal procedures and requirements. All financial sources are fully and clearly stated in the acknowledgments section. A signed document has been filed in the journal archives.

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