Diversity patterns of alien and native plant species in Trieste port area: exploring the role of urban habitats in biodiversity conservation

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Abstract Nowadays, urban areas play a crucial role in biodiversity conservation and habitat protection despite the constant pressures on which these habitats are subjected. They may even host relatively new plant communities due to the peculiar ecosystem where they vegetate. The port of Trieste (NE Italy) is characterized by a mixed mosaic of intensely human impacted areas (where commercial activities are still ongoing) flanked by abandoned areas where vegetation persists or has spontaneously recovered. In this study, we sampled the whole port area through a stratified random sampling by placing multiscalar nested plots in four different habitats (*strata*) previously identified by photo-interpretation. Plant species richness and abundance were assessed in each plot. Each species was then classified as native or alien and patterns of species richness and complementarity were compared among habitats. Results show that there is a significant difference in species richness patterns among habitats, while observed patterns are likely to vary at different spatial scales. As expected, urban plots account for most of the alien species in the sampling, while wooded plots cope better with invasion, accounting for a lower alien/native ratio. These results highlight how habitat diversity enhances biodiversity in urban areas and how it could provide an effective filtering effect able to reduce the spread of alien species. In addition, we provide further evidence for the use of multi-scale approaches in order to study the complex relationships between spatial heterogeneity and plant species richness.

Introduction

Nowadays, urbanization is occurring at a rapid pace being deemed as one of the main causes of biotic homogenization (McKinney 2006; Kühn and Klotz 2006), also affecting local alpha and beta-diversity (Socolar et al. 2016). Urban ecosystems evolve over time and space, emerging as the outcome of dynamic interactions between socio-economic and biophysical processes operating over multiple scales. To date, preserving large, intact areas of natural habitat is a key means for preserving biodiversity. However, this may not be feasible in highly urbanized territories, where there is little natural habitat available. For a long time, urban areas have been seen as degraded environments, characterized by species of low ecological value where natural ecosystems are usually absent (Guneralp and Seto 2013). However, in the last decade, several studies demonstrate how small and fragmented natural habitats can persist in highly human-affected areas, accounting for relatively high levels of biodiversity in terms of species richness (Kühn et al. 2004; Cornelis and Hermy 2004; Alvey 2006; Aronson et al. 2015). It has already been highlighted how urban environment may host species of local or regional conservation value (Rebelo et al. 2011; Vähä-Piikkiö et al. 2004) and even red-listed plant species (Gustafsson 2002). A significant variation

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in these patterns has been observed both across cities (Kühn et al. 2004; Aronson et al. 2014) than within cities (Sushinsky et al. 2013; Beninde et al. 2015). Urban ecosystems are characterized by a high heterogeneity, which is attributable to the presence, in a small extent, of different land uses (Savard et al. 2000; Ortega-Álvarez and MacGregor-Fors 2009). Moreover, it has been observed that some taxa are able to form atypical assemblages in cities, as species which are typical of different ecosystems coexist in an "artificial" environment (McKinney 2006). Consequently, these remnants can be easily considered as "hotspots" for species richness (sensu Knapp et al. 2008; Croci et al. 2008) and, accordingly, for biodiversity conservation compared to their rural surroundings. Nevertheless, human-altered environments, such as cities, are more often the introduction epicenters for alien plant species, in particular for Invasive Alien Species (hereafter IAS, Vitousek et al. 1997; Pyšek 1998). High human pressure and the contemporary presence of numerous pathways of introduction and spread (e.g. transport, human activities) alongside other factors such as "heat island effect", ensure a high dispersion rate of the propagules and the reduction of competition with native species, making these ecosystems prone to biological invasions (Hulme 2003; Chytrý et al. 2008; Bacaro et al. 2015). Within urban environments, port areas represent preferential places from where alien species move into new ecosystems, as shipping routes represent one of the main pathways of introduction and colonization (Molnar et al. 2008; Adhikari et al. 2015). The IAS in port areas have been extensively explored with regard to the introduction of marine species, especially fish and jellyfish (e.g. Galil 2000; Bax et al. 2003) while few descriptions of patterns and distributions of plant IAS are reported in these peculiar areas.

Considering the strong dependence between biodiversity metrics such as species richness and spatial scale (Wiens 1989; Crawley and Harral 2001; Stohlgren 2007), a multiscalar approach should be applied in the analysis of biodiversity patterns. As far as we know, this relationship is not constant and varies with both spatial grain (the size of the sampling unit) and spatial extent (the maximum distance among them, or simply the extension of the study area), as pointed out by Rahbek (2005), among others. In the last two centuries, Trieste (NE Italy) has developed behind its harbor, wedged in the foothills of the Karst plateau. The singularity of Trieste's port area lies in its conformation. It is a mixed mosaic of intensely human impacted areas (where commercial activities are still ongoing) flanked by abandoned patches, where vegetation has been naturally restored. For these reasons, it can be considered an interesting environment where to assess biodiversity patterns within urban area, with a special focus on alien species component of plant communities. As a consequence, the main aims of this paper are: i) to investigate the plant diversity in the Trieste port area through quantitative methods; ii) to explore diversity patterns of native and alien species and assess their relationships within this environment. Both objectives aim at describing the important role of urban ecosystems and their mosaic as biodiversity hotspots, as well as highlighting their possible role in the context of plant invasion mechanisms.

Materials and methods

Study area

Vegetation data used in this study were collected in the port area of Trieste (centroid coordinates: 45.6640° N-13.7987° E, *datum*: WGS 84); the whole sampling area covers a surface of 3.48 km^2, being located at sea level (Fig. 1). The main natural vegetation types recognizable in the surveyed green areas of the port may be attributed to a Mediterranean scrubland. They correspond to the phytosociological class *Quercetea ilicis* in the northernmost portion, while in the southern part, which is characterized mostly by marine clays and silts, there are remnants of an old wetland, which was reclaimed after the Second World War to build part of the current industrial area of Trieste. Here it is possible to find fragments of the typical vegetation of salt marshes (closer to the sea) and swamps, which belong to classes *Juncetea maritimi* and *Phragmitetea,* although plant communities are severely altered by the high anthropic pressure and the strong modifications of habitats and soils.

Sampling design and data collection

Vegetation data were collected by means of a probabilistic sampling. Specifically, the sampling design was based on a hierarchical stratified random sampling. At first, manual photo-interpretation and digitalization of the whole port area was performed using regional orthophotos (year 2011, spatial resolution 50 cm, source IRDAT FVG). Then the sampling area was classified by habitat type (four classes: urban, meadow, shrubland and woodland), later verified on the field. The urban habitat encompasses those environments that are typical of cities, such as roadsides, railway lines, brownfields and all those niches where human pressure is higher. The identification of the other three habitats was based on the percentage of the vegetation cover (later verified on the field). The selection of sampling units worked as follows: firstly, a grid of 1 × 1 km was superimposed to habitat map of the port area. Subsequently, a total of 100 random sampling units (plots) were selected, with the number of sampling units being proportional to the area of habitat types in each grid cell. This sampling procedure ensures a representative coverage of the whole territory, avoiding sampling points to be clustered. Each sampling unit was identified in the field by means of a high precision GPS (Leica Nexteq T6, accuracy <2 m). A nested

Fig. 1 Satellite image representing the city of Trieste (right part). Black line represents the perimeter of the study area (Port Area)
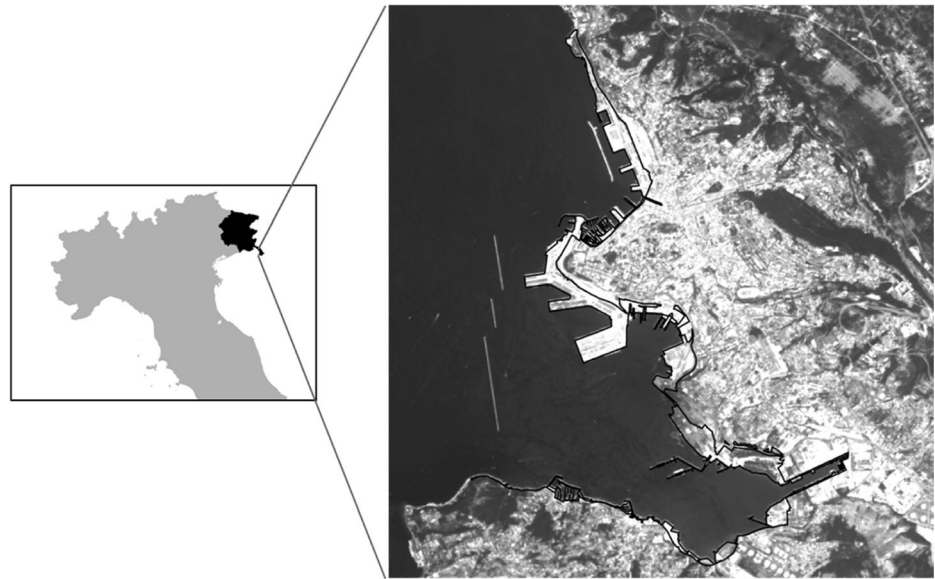
multiscalar plot with four spatial scales (linear dimension: 0.5 m, 1 m, 2 m, 4 m) was then materialised in the field and the vegetation was sampled. The complete inventory of plant species occurring in each sampling unit was listed for each plot scale along with the percentage coverage of each species (visual estimate) for the largest spatial scale (16 m^2). Data were collected during the period June–September 2015. Nomenclature and taxonomy follow Pignatti (1982) and Conti et al. (2005). All species were classified as native or alien, according to the inventory of the Italian alien flora (Celesti-Grapow et al. 2009). Moreover, alien species were further classified into archaeophytes and neophytes (introduced before or after ca. 1500, respectively; Pyšek et al. 2004).

Analysis of plant diversity patterns for alien and native species

Differences in plant species richness were statistically evaluated for each plot scale and habitat using non-parametric Kruskal-Wallis rank sum test. Where the test resulted significant, an adjusted *posteriori* pairwise comparison was performed between pairs of habitats (using the "kruskalmc" function in the R package "pgirmess"; Giraudoux 2016).

Plot-based rarefaction curves (hereafter SAC, Gotelli and Colwell 2001) were computed at each spatial scale, considering the whole dataset and native and alien species separately. The analytic solution proposed by Kobayashi was used (1982, for details see Chiarucci et al. 2008) using the R "vegan" package (Oksanen et al. 2016). For an analysis of spatial autocorrelation in the species distribution patterns, SAC were compared with Spatially-Constrained Rarefaction curves (hereafter SCR, Chiarucci et al. 2009; Bacaro et al. 2012) using the R code provided in Bacaro et al. (2012) which

allows to consider spatial autocorrelation of the samples in the calculation of rarefaction curves. This recent methodology proved to be more effective in comparing biodiversity patterns among areas with different extent (Bacaro et al. 2016). The ratio between the plot-based rarefaction curve of alien and native species was finally calculated and used to describe how patterns of these two groups vary among plots for an increasing sampling effort. All spatial scales were analyzed in this way.

Species diversity patterns were also compared using additive partitioning techniques (Lande 1996; Gering et al. 2003; Crist et al. 2003): partition of alpha, beta and gamma diversity elements across different scales (plot scale, habitat scale and whole sampling area) was performed for each group (native and alien species). Each of these diversity component was expressed as a proportion of the total species richness. For calculation of diversity elements at different spatial scales, samples were nested from lower hierarchical level within higher units, so that the spatial component increases constantly from $i = 1$ to $i = l$, where l is the spatial scale investigated. Species richness in the whole sampling area was partitioned into the inventory diversities at the various spatial scales (α plot, α habitat), that added up to the differentiation diversities for the corresponding spatial scales (β plot, β habitat), give the total diversity of the whole sampling area (γ). According to the above, α_l reflects the average diversity found within plots. Values at higher sampling levels have been obtained as:

$$\beta_l = \gamma - \alpha_l \qquad (1)$$

where l are the levels of sampling with samples in lower hierarchical levels nested within higher level units. For each lower sampling level as:

$$\beta_i = \alpha_{i+1} - \alpha_i \qquad (2)$$

Finally, additive partition was calculated as:

$$\gamma = \alpha_i + \sum_{j=i}^{l} \beta_j \qquad (3)$$

Departures from random expectations of species richness values were evaluated through null model testing, according to a nested hierarchical sampling design. 999 permutations of the raw matrix were performed to assess significance in the deviation from randomness in the observed patterns.

In term of species assemblages, plot-to-plot differences in species composition for the whole recorded set of species and for the alien group alone were analyzed using the Non-metric MultiDimensional Scaling procedure (hereafter NMDS; Kruskal and Wish 1978); before performing NMDS, abundance data were log-transformed and the Bray-Curtis dissimilarity metric was calculated. All statistical analyses were conducted using R 3.3.2 (R Core Team 2016) except NMDS, that was computed using Primer 6 software (Clarke and Warwick 2005).

Results

In total, 234 species were collected: 203 native species (86.75% of the whole sample) and 31 alien species, corresponding to 13.25% of the total species richness (according to Celesti-Grapow and Accogli 2010, the Italian national average is 13.4% whereas the percentage for the Friuli Venezia Giulia Region is 11.6%). Mean species richness values at different spatial scales are shown in Table 1. The most represented Families were *Poaceae* (18%), *Asteraceae* (17%), *Fabaceae* (12%) and *Rosaceae* (7%). *Dactylis glomerata* subsp. *glomerata* was the most frequent species (62% of the sampled plots), followed by *Rubus ulmifolius* (60%) and *Daucus carota* subsp. *carota* (55%). Among alien species, the most abundant were *Sorghum halepense* (27%), *Robinia pseudoacacia* (26%), *Senecio inaequidens* (25%) and *Ambrosia artemisiifolia* (18%). Neophytes comprise 26 taxa (11.11%) and Archaeophytes 5 taxa (2.14%). Most aliens have American (53.33% of the total alien flora) and Asiatic origin (30%). Table 2 shows proportional representations of aliens, archaeophytes, neophytes and their ratio in city floras of Italy for which published data are available.

Species richness strongly varied according to habitat, group and spatial scale (Table 3). For the whole species pool, comparisons of the mean ranks between habitats at larger scale showed that species richness is not significantly different ($\alpha = 0.05$) when urban habitat is compared to meadow (observed rank mean difference = 17.8; $p > 0.05$). Conversely, statistical differences were observed when woodlands and

Table 1 Species richness (Mean ± SD) at each spatial scale considering all species pooled and for native and alien species separately

Species group	Plot linear dimension			
	0.5 m	1 m	2 m	4 m
Total	4.4 ± 2.2	6.5 ± 3.2	9.1 ± 4.2	13.5 ± 5.7
Native	3.85 ± 2.1	5.68 ± 3.0	7.9 ± 4.0	11.7 ± 5.3
Alien	0.54 ± 0.7	0.81 ± 0.9	1.2 ± 1.1	1.8 ± 1.4

shrublands were compared to meadow (observed mean difference woodlands vs meadows = 17.27, $p < 0.05$; shrubland vs meadow =19.12, $p < 0.05$). At the smallest spatial scale (0.25 m^2) Kruskal-Wallis resulted significant (χ^2 (3) = 10, $p = 0.01$) with meadow vs urban habitat showing significant differences (observed rank mean difference = 26.66; $p < 0.05$). No differences were observed among habitats at the intermediate spatial scales (1 and 4 m^2). With regard to native species, we observed a significant result just at the smaller spatial scale where there is a difference among plots in meadow and urban habitat (observed rank mean difference = 27; $p < 0.05$). Alien species show significant differences outputs just at 16 m^2 and 4 m^2 and in both cases the difference was detected between woodland and meadow (observed rank mean difference is 23.4 and 19.6, $p < 0.05$, respectively). The highest values of species richness, considering all species pooled, have been detected in meadows and woodlands, and the same pattern has been confirmed also for native species. Conversely, alien species tend to have higher species richness in urban plots as expected even though meadows show similar values as well. SCR and SAC present similar patterns for each spatial scale (see Supplementary materials for a comparison between SCRs and the classic rarefaction curves). The lack of any asymptotic trend characterizes all curves. The spatial scale of analysis does not affect patterns in species accumulation even when native and alien species are analyzed separately (Fig. 2), although SAC for alien species reach an asymptotic trend, which means that, on average the majority of the alien species pool in the sampled area have been sampled. When rarefactions were calculated for each habitat considering all the species pooled, observed patterns slightly differ according to scale (Fig. 3). In particular, it is worth noting as the woodland curve lies above the others at larger scales (but below at lower scales) and the urban curve is higher than shrubland curve at smaller scales. All SCR curves fall below the SAC (see Supplementary materials for more details), indicating that plant communities are spatially autocorrelated despite habitats being prone to fragmentation effects. The ratio between native and alien rarefactions results in slightly different outputs: this pattern varies among scales even though a general trend can be observed (Fig. 4). At the smallest scale, we can observe a smaller ratio that can be explained by the fact that less alien species are sampled by chance, given their low number in

Table 2 Proportions of alien species, archaeophytes, neophytes and their ratio in city floras of the Italian peninsula for which published data exist. (Al: Alien, Arch: Archaeophytes, Neo: Neophytes)

City	Population	% Al	% Arch	% Neo	Ratio	Source
Ancona	101,331	11.9	4	7.9	0.51	Celesti-Grapow and Blasi (1998)
Cagliari	154,422	11.9	4.3	7.6	0.57	Celesti-Grapow and Blasi (1998)
Milano	1,344,906	25.6	4.2	21.4	0.20	Celesti-Grapow and Blasi (1998)
Palermo	674,565	14.3	4.7	9.6	0.49	Celesti-Grapow and Blasi (1998)
Roma	2,867,143	12.4	4.3	8.3	0.52	Celesti-Grapow and Blasi (1998)
Trieste	204,419	13.36	2.26	11.10	0.20	Martini and Altobelli (2009)
Trieste (port area cells grid)	-	15.10	2.56	12.54	0.20	Martini and Altobelli (2009)
Trieste (port area)	-	13.25	2.14	11.11	0.19	Present study

proportion to native species. On an increasing plot scale, a higher ratio occurred, with values more closely related to the actual rate of invasion expected for the region.

Diversity element proportion (α and β) increased according to increasing grain sizes. All calculated values were significantly different from random expectations, meaning that these elements are highly spatially structured. When considered separately, native and alien diversity patterns were constant across scales (see Supplementary materials). The partition of diversity according to habitat (Fig. 5) shows that urban environment accounts the highest proportion of alien species diversity at the plot level (α plot) for each scale investigated; on the contrary, woodland and shrubland habitats have the smallest mean percentage in aliens. It is worth noting how urban habitat hosts a high proportion of native species compared to other environments considered in this study. The NMDS analysis for native species provides a fairly good representation of the data (Fig. 6a, stress =0.23): it can be noted as urban plots set aside respect to the others, indicating their compositional differentiation. NMDS computed only on alien species (Fig. 6b, stress =0.15) clearly shows the absence of specific pattern among different habitats, highlighting how the urban ecosystem may be prone to a diffuse propagule pressure by alien species, resulting in alien flora homogenization across the whole studied area. In addition, it provides further evidence of the high level of habitat plasticity that alien species are able to tolerate, thanks to their great adaptability to various types of environment.

Table 3 Output of the Kruskal Wallis test at each spatial scale for each species group (*** = $p < 0.001$; ** = $p < 0.01$; * = $p < 0.05$; ns = not significant)

Species Group	Plot linear dimension			
	4 m	2 m	1 m	0.5 m
Total	$\chi^2(3) = 9$*	$\chi^2(3) = 5$ ns	$\chi^2(3) = 8$ ns	$\chi^2(3) = 10$**
Native	$\chi^2(3) = 7$ ns	$\chi^2(3) = 4$ ns	$\chi^2(3) = 7$ ns	$\chi^2(3) = 10$**
Alien	$\chi^2(3) = 10$**	$\chi^2(3) = 10$*	$\chi^2(3) = 3$ ns	$\chi^2(3) = 1$ ns

Discussion

The role of green areas in urban ecosystem

Green areas inside cities play a key role in global efforts to protect and manage vulnerable ecosystems and biodiversity (Goddard et al. 2010). The correct management and conservation of these habitats may mitigate the rate of biodiversity loss and habitat destruction (Alvey 2006). Furthermore, they may provide benefits in terms of ecosystem services (e.g. Brack 2002). Here, we quantitatively evaluated plant diversity elements as well as their patterns in the context of the urban habitat mosaic induced by human activities in a peculiar area: the port of Trieste. The effect of the habitat on species richness has long been examined and it is well known to influence both natural and urban environments (Pyšek et al. 2002; Celesti-Grapow et al. 2006). Knapp et al. (2008) proved how plant species vegetating in urban environments are endowed with those functional traits that make them well suited to cope with stresses. In fact, they are usually wind-pollinated, zoochores and with scleromorphic leaves. As already reported by others authors (Deutschewitz et al. 2003; Kühn et al. 2004), these environments are primarily colonized by many R-strategy native species (Ruderals sensu Grime's CSR classification, Grime et al. 1988) that could contribute to increase local alpha diversity. These species are generally highly resistant to disturbances (intended as the mechanism which limits plant biomass by causing its partial or total destruction, Grime 2006), and tend to have short life-spans (generally annual or short-lived perennial), early flowering with a large amount of seeds (Grime 2006), all features that make these species well-adapted to urban ecosystem. Our findings mirror these studies very well; furthermore, we highlighted that the urban habitat shows high proportions of alpha diversities both in native and alien species compared to more natural habitats such as woodlands or shrublands, even though its flora may be primarily constituted by alien species, especially where urban heat island effect is more pronounced (McKinney 2006). Thus, we expected to find a high proportion of alien species, as already

Fig. 2 Spatially-Constrained Rarefaction curves (SCR) and classical rarefaction curves (SAC) comparing native and alien species for each spatial scale investigated
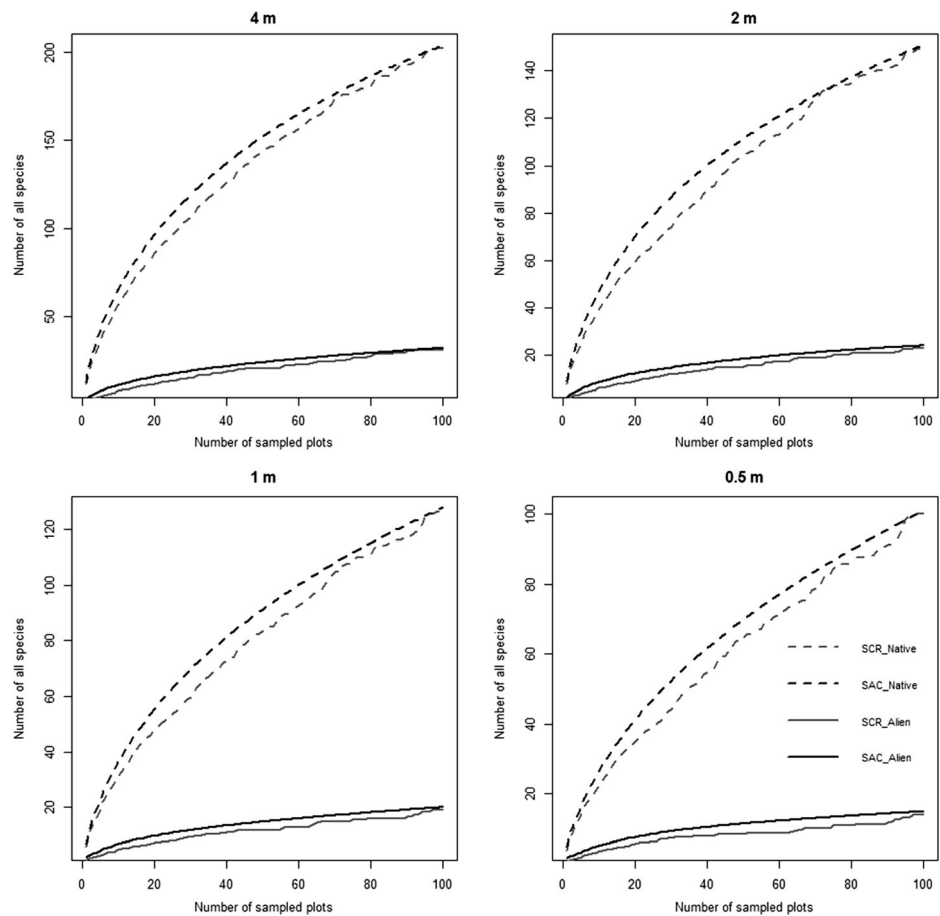
demonstrated in many other studies (Ricotta et al. 2010; Kowarik et al. 2013; Aronson et al. 2015 among others). Indeed, the constant availability of propagules and the great connectivity present within cities ensure the maintenance of established populations. Additionally, other factors such as landscape fragmentation, heat island effect, and the presence of hard surface that promote aridity due to run off, create conditions that allow the establishment and spread of those species, like alien plants, that are ecologically pre-adapted to cope with stresses and harsh conditions. On the other hand, the low alien percentages found in woodland and shrubland may be explained according to the theory of biotic resistance (Levine 2000). This states that resident species in a community reduce the success of exotic invasion (Levine et al. 2004) through biotic filters that may impede plant invasion such as competition from native species (mainly through shading effect considering that most of alien plants are sun-loving), pathogens and herbivores.

The high steepness of the rarefaction curves suggests that further sampling efforts would have been necessary to capture the complete species pool of native species, conversely to what was observed for alien species. The ratio among rarefaction curves is an interesting tool to disentangle the relative contribution of typical groups such as alien vs native species

(e.g. Pyšek et al. 2002). The high proportion of neophytes to the detriment of archaeophytes combined with the low archaeophytes to neophytes ratio (compared to other cities, see Table 2) demonstrates how in Trieste alien species have spread in relatively recent times. Our results concur with those obtained by Martini and Altobelli (2009) in a census of the flora of the city of Trieste, both considering the whole urban area and the grid cells only which encompass the port area. This allows us to make some considerations: firstly, that probabilistic sampling is efficient (in term of both costs and time) in capturing actual diversity patterns and plant variability within urban areas; secondly, that the rate of invasion in Trieste is very high, considering its small population and extension compared to other Italian cities for which data are available. This could be probably explained chiefly by the role of port areas as preferential pathway of introduction and spread of alien species; this is further confirmed by the very low proportion of the more ecologically specialized archaeophytes compared to other Italian cities.

Effects of scale on plant species richness

This study points out how assessing species richness with a multi-scale approach is desirable in order to study the complex

Fig. 3 SCR curves for all species pooled divided according to habitat type for each spatial scale investigated. The difference in expected species number for number of plot between the two curves ranged from fifteen to eight reducing the scale of observation, all these patterns are constant across scales
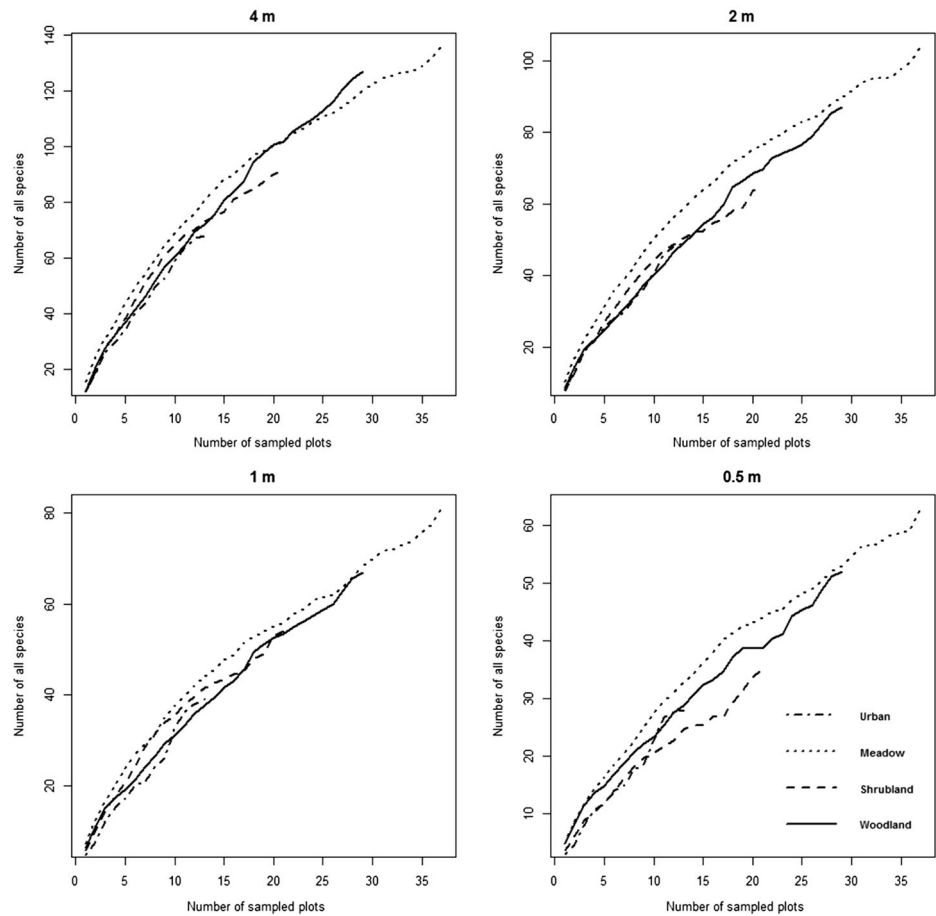
relationships between spatial scale and plant species richness at different grain sizes. Similar evidences have been demonstrated by others authors in natural contexts (Crawley and Harral 2001; Palmer et al. 2006 for N-American flora; Chiarucci et al. 2012 for a network of protected areas). Our
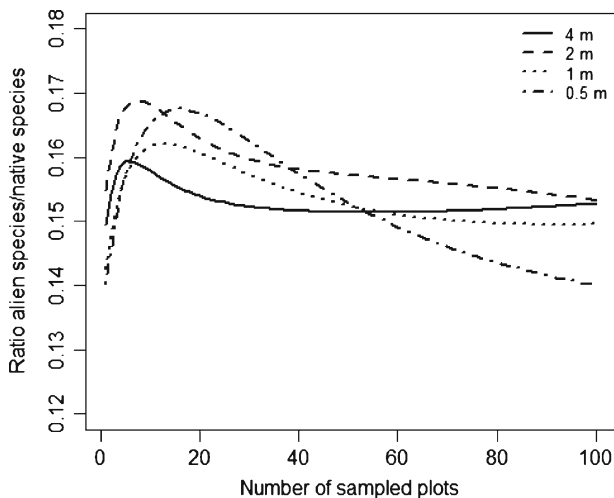

Fig. 4 Ratio between rarefaction curves calculated for alien and native species at each spatial scale investigated

results suggest the need to use different spatial scales to detect a reliable rate of invasion in an area, since small scales are not able to capture the real amount of alien species due to sampling effects. Spatial heterogeneity is considered as the major driver in shaping species composition (Davies et al. 2005); this encompasses several factors such as ecological interactions mainly at the local scale, or geography-based factors such as topography or aspect on a larger scale (Fridley et al. 2004). Since urban development patterns affect spatial heterogeneity of urban ecosystems, we also argue that alternative urban patterns that emerge from human and ecological interactions play an important role in the dynamics and resilience of these peculiar areas.

Conclusion

Cities are highly anthropogenic ecosystems, further challenging the ecological understanding of how novel species assemblages support urban ecosystem services. As urban areas continue to expand in the coming decades, so will the size of the urban green habitat: however, the role of particular groups of species (native, alien, invasive) in the supply and demand for ecosystem services remains poorly understood (Haase et al. 2014).

Fig. 5 Alpha and beta components of diversity (% of the total) for native and alien species for each habitat on a) 4 × 4 m, b) 2 × 2 m, c) 1 × 1 m, d) 0.5 × 0.5 m. The contributions to the total richness for each scale were determined by the additive partitioning of diversity method (*** = $p < 0.001$; ** = $p < 0.01$; * = $p < 0.05$; NS = Not Significant)
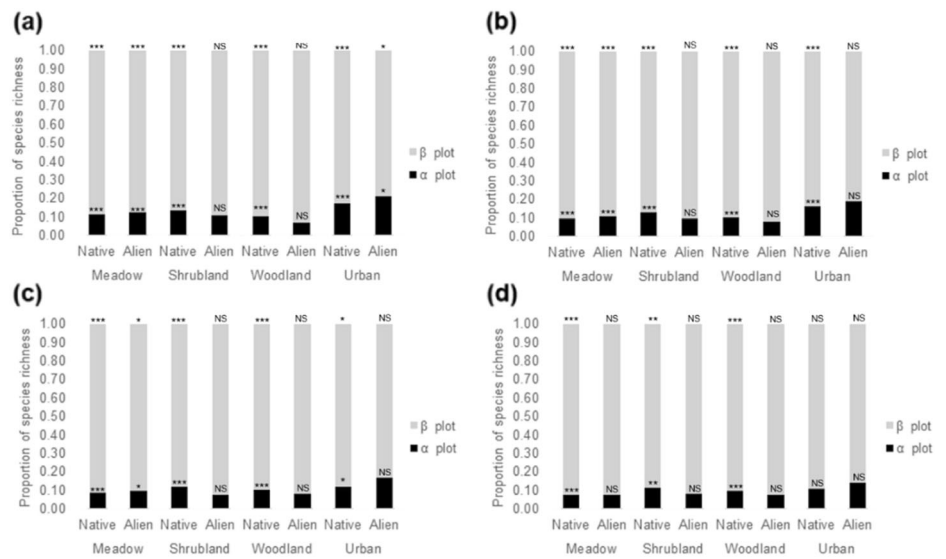
Additionally, understanding habitat function and habitat connectivity is a key target for city planners to design appropriate management and conservation strategies for urban biodiversity and ecosystem resilience. To our knowledge, this is one of the first attempts to explore plant diversity patterns in port areas: this evaluation assumes particular importance, since ports are deemed one of the main pathways for alien species introduction and spread.

Our findings contribute to integrate an increasing literature about port floras shedding light on the role of port areas as potential hotspots for biodiversity conservation. We also explored the habitat-filtering role of green areas with respect to

Fig. 6 NMDS based on Bray-Curtis dissimilarity matrix (log-transformed abundance data) for the largest spatial scale (16 m²) for each habitat. **a**) Native species **b**) Alien species
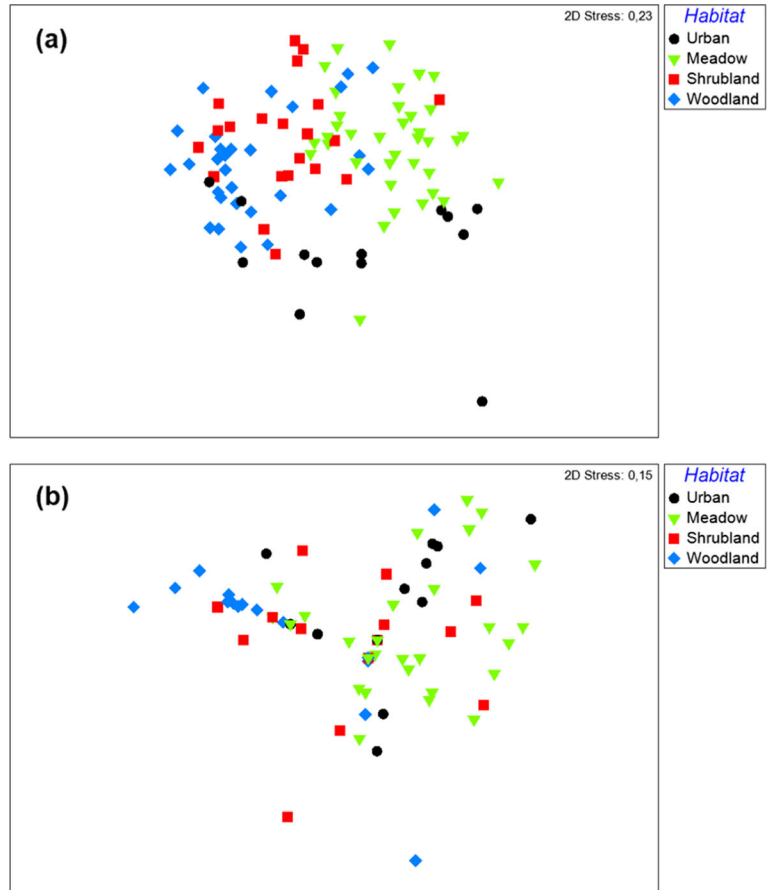
the spread of alien plant species and the prominent role that green areas seems to have in tackling alien plant invasion inside urban environment. These remnants of natural vegetation are fundamental to preserve native vegetation and ecosystem services, mainly considering the ongoing pressures occurring in our ecosystems such as urban sprawl, globalization and global warming. Under this scenario, alien species should be favored, and their spread and establishment may be enhanced; for these reasons, maintaining large green areas inside urban ecosystem could preserve local biodiversity against an increasing alien species pressure. Future investigations and sampling efforts are needed to better understand how native and alien species interact in urban habitats, specifically, it could be interesting to develop a system of horizon scanning (e.g. Sutherland and Woodroof 2009) for risk assessment and management, to foresee the arrival of new alien species, for instance through the study of the main shipping routes in port areas. Furthermore, the application of a multi-scale integrated approach in vegetation analysis may provide useful ecological insights to understand better these ecosystems, allowing the development of enhanced management strategies. There is a hope that an increased knowledge of how urban plant communities are structured will allow us to design cities that promote biodiversity and the survival of native plant species pool.

References

Adhikari D, Tiwary R, Barik SK (2015) Modelling hotspots for invasive alien plants in India. PLoS One 10(7):e0134665. doi:10.1371/journal.pone.0134665

Alvey AA (2006) Promoting and preserving biodiversity in the urban forest. Urban For Urban Gree 5(4):195–201. doi:10.1016/j.ufug.2006.09.003

Aronson MFJ, Handel SN, La Puma IP, Clemants SE (2015) Urbanization promotes non-native woody species and diverse plant assemblages in the New York metropolitan region. Urban Ecosys 18:31–45. doi:10.1007/s11252-014-0382-z

Aronson MFJ, La Sorte FA, Nilon CH, Katti M, Goddard MA, Lepczyk CA, Warren PS, Williams NSG, Cilliers S, Clarkson B, Dobbs C, Dolan R, Hedblom M, Klotz S, Louwe Kooijmans J, Kühn I, MacGregor-Fors I, McDonnell M, Mörtberg U, Pyšek P, Siebert S, Sushinsky J, Werner P, Winter M (2014) A global analysis of the impacts of urbanization on bird and plant diversity reveals key anthropogenic drivers. Proc R Soc B 281:20133330. doi:10.1098/rspb.2013.3330

Bacaro G, Altobelli A, Cameletti M, Ciccarelli D, Martellos S, Palmer MW, Ricotta C, Rocchini D, Scheiner SM, Tordoni E, Chiarucci A (2016) Incorporating spatial autocorrelation in rarefaction methods: implications for ecologists and conservation biologists. Ecol Indic 69:233–238. doi:10.1016/j.ecolind.2016.04.026

Bacaro G, Maccherini S, Chiarucci A, Jentsch A, Rocchini D, Torri D, Gioria M, Tordoni E, Martellos S, Altobelli A, Otto R, Escudero CG, Fernández-Lugo S, Fernández-Palacios JM, Arévalo JR (2015) Distributional patterns of endemic, native and alien species along a roadside elevation gradient in Tenerife, Canary Islands. Community Ecol 16(2):223–234. doi:10.1556/168.2015.16.2.10

Bacaro G, Rocchini D, Ghisla A, Marcantonio M, Neteler M, Chiarucci A (2012) The spatial domain matters: spatially constrained species rarefaction in a free and open source environment. Ecol Complex 12:63–69. doi:10.1016/j.ecocom.2012.05.007

Bax N, Williamson A, Aguero M, Gonzalez E, Geeves W (2003) Marine invasive alien species: a threat to global biodiversity. Mar Policy 27(4):313–323. doi:10.1016/S0308-597X(03)00041-1

Beninde J, Veith M, Hochkirch A (2015) Biodiversity in cities needs space: a meta-analysis of factors determining intra-urban biodiversity variation. Ecol Lett 18(6):581–592. doi:10.1111/ele.12427

Brack CL (2002) Pollution mitigation and carbon sequestration by an urban forest. Environ Pollut 116:195–200. doi:10.1016/S0269-7491(01)00251-2

Celesti-Grapow L, Accogli R (2010) Flora vascolare alloctona e invasiva delle regioni d'Italia. Università degli Studi di Roma La Sapienza

Celesti-Grapow L, Alessandrini A, Arrigoni PV, Banfi E, Bernardo L, Bovio M, Brundu G, Cagiotti MR, Camarda I, Carli E, Conti F, Fascetti S, Galasso G, Gubellini L, La Valva V, Lucchese F, Marchiori S, Mazzola P, Peccenini S, Poldini L, Pretto F, Prosser F, Siniscalco C, Villani MC, Viegi L, Wilhalm T, Blasi (2009) Inventory of the non-native flora of Italy. Plant Biol 143(2):386–430. doi:10.1080/11263500902722824

Celesti-Grapow L, Blasi C (1998) A comparison of the urban flora of different phytoclimatic regions in Italy. Glob Ecol Biogeogr 7:367–378. doi:10.1046/j.1466-822x.1998.00304

Celesti-Grapow L, Pyšek P, Jarošík V, Blasi C (2006) Determinants of native and alien species richness in the urban flora of Rome. Divers Distrib 12:490–501. doi:10.1111/j.1366-9516.2006.00282.x

Chiarucci A, Bacaro G, Filibeck G, Landi S, Maccherini S, Scoppola A (2012) Scale dependence of plant species richness in a network of protected areas. Biodivers Conserv 21(2):503–516. doi:10.1007/s10531-011-0196-8

Chiarucci A, Bacaro G, Rocchini D, Fattorini L (2008) Discovering and rediscovering the sample-based rarefaction formula in the ecological literature. Community Ecol 9:121–123. doi:10.1556/ComEc.9.2008.1.14

Chiarucci A, Bacaro G, Rocchini D, Ricotta C, Palmer M, Scheiner SM (2009) Spatially constrained rarefaction: incorporating the autocorrelated structure of biological communities into sample-based rarefaction. Community Ecol 10(2):209–214. doi:10.1556/10.1556/ComEc.10.2009.2.11

Chytrý M, Jarošík V, Pyšek P, Hájek O, Knollová I, Tichý L, Danihelka J (2008) Separating habitat invasibility by alien plants from the actual level of invasion. Ecology 89(6):1541–1553. doi:10.1890/07-0682.1

Clarke KR, Warwick RM (2005) Primer-6 computer program. Natural Environment Research Council, Plymouth

Conti F, Abbate G, Alessandrini A, Blasi C (2005) An annotated checklist of the Italian vascular Flora. Palombi Ed, Roma

Cornelis J, Hermy M (2004) Biodiversity relationships in urban and suburban parks in Flanders. Landscape Urban Plan 69(4):385–401. doi:10.1016/j.landurbplan.2003.10.038

Crawley MJ, Harral JE (2001) Scale dependence in plant biodiversity. Science 291(5505):864–868. doi:10.1126/science.291.5505.864

Crist TO, Veech JA, Gering JC, Summerville KS (2003) Partitioning species diversity across landscapes and regions: a hierarchical analysis of α, β, and γ-diversity. Am Nat 162:734–743. doi:10.1086/378901

Croci S, Butet A, Georges A, Aguejdad R, Clergeau P (2008) Small urban woodlands as biodiversity conservation hot-spot: a multi-taxon approach. Landscape Ecol 23(10):1171–1186. doi:10.1007/s10980-008–9257-0

Davies KF, Chesson P, Harrison S, Inouye BD, Melbourne BA, Rice KJ (2005) Spatial heterogeneity explains the scale dependence of the native-exotic diversity relationship. Ecology 86(6):1602–1610. doi:10.1890/04-1196

Deutschewitz K, Lausch A, Kühn I, Klotz S (2003) Native and alien plant species richness in relation to spatial heterogeneity on a regional scale in Germany. Glob Ecol Biogeogr 12(4):299–311. doi:10.1046/j.1466-822X.2003.00025.x

Fridley JD, Brown RL, Bruno JF (2004) Null models of exotic invasion and scale-dependent patterns of native and exotic species richness. Ecology 85(12):3215–3222. doi:10.1890/03-0676

Galil BS (2000) A sea under siege–alien species in the Mediterranean. Biol Invasions 2(2):177–186. doi:10.1023/A:1010057010476

Gering JC, Crist TO, Veech JA (2003) Additive partitioning of species diversity across multiple spatial scales: implications for regional conservation of biodiversity. Conserv Biol 17:488–499. doi:10.1046/j.1523-1739.2003.01465.x

Giraudoux P (2016). Pgirmess: data analysis in ecology. R package version 1.6.4.

Goddard MA, Dougill AJ, Benton TG (2010) Scaling up from gardens: biodiversity conservation in urban environments. Trends Ecol Evol 25(2):90–98. doi:10.1016/j.tree.2005.05.013

Gotelli NJ, Colwell RK (2001) Quantifying biodiversity: procedures and pitfalls in the measurement and comparison of species richness. Ecol Lett 4(4):379–391. doi:10.1046/j.1461-0248.2001.00230.x

Grime JP (2006) Plant strategies, vegetation processes, and ecosystem properties. John Wiley & Sons

Grime JP, Hodgson JG, Hunt R (1988) Comparative plant ecology: a functional approach to common British species. Chapman & Hall, London, UK

Guneralp B, Seto KC (2013) Futures of global urban expansion: uncertainties and implications for biodiversity conservation. Environ Res Lett 8:014025. doi:10.1088/1748-9326/8/1/014025

Gustafsson L (2002) Presence and abundance of red-listed plant species in Swedish forests. Conserv Biol 16:377–388 http://www.jstor.org/stable/3061364

Haase D, Larondelle N, Andersson E, Artmann M, Borgstrom S, Breuste J et al (2014) A quantitative review of urban ecosystem service assessments: concepts, models, and implementation. Ambio 43:413–433. doi:10.1007/s13280-014-0504-0

Hulme PE (2003) Biological invasions: winning the science battles but losing the conservation war? Oryx 37(2):178–193. doi:10.1017/S003060530300036X

Knapp S, Kühn I, Schweiger O, Klotz S (2008) Challenging urban species diversity: contrasting phylogenetic patterns across plant functional groups in Germany. Ecol Lett 11(10):1054–1064. doi:10.1111/j.1461-0248.2008.01217.x

Kobayashi S (1982) The rarefaction diversity measurement and the spatial distribution of individuals. Jpn J Ecol 32:255–258

Kowarik I, Lippe M, Cierjacks A (2013) Prevalence of alien versus native species of woody plants in berlin differs between habitats and at different scales. Preslia 85(2):113–132

Kruskal JB, Wish M (1978) Multidimensional scaling, Sage University paper series on quantitative application in the social Sciences, 07–011. Sage Publications, Beverly Hills and London

Kühn I, Brandl R, Klotz S (2004) The flora of German cities is naturally species rich. Evol Ecol Res 6(5):749–764

Kühn I, Klotz S (2006) Urbanization and homogenization comparing the floras of urban and rural areas in Germany. Biol Conserv 127:292–300. doi:10.1016/j.biocon.2005.06.033

Lande R (1996) Statistics and partitioning of species diversity, and similarity among multiple communities. Oikos 76:5–13. doi:10.2307/3545743

Levine JM (2000) Species diversity and biological invasions: relating local process to community pattern. Science 288:852–854. doi:10.1126/science.288.5467.852

Levine JM, Adler PB, Yelenik SG (2004) A meta-analysis of biotic resistance to exotic plant invasions. Ecol Lett 7:975–989. doi:10.1111/j.1461-0248.2004.00657.x

Martini F, Altobelli A (2009) Flora vascolare spontanea di Trieste. Lint, Trieste

McKinney ML (2006) Urbanization as a major cause of biotic homogenization. Biol Conserv 127:247–260. doi:10.1016/j.biocon.2005.09.005

Molnar JL, Gamboa RL, Revenga C, Spalding MD (2008) Assessing the global threat of invasive species to marine biodiversity. Front Ecol Environ 6:485–492. doi:10.1890/070064

Oksanen J, Blanchet GF, Kindt R, Legendre P, Minchin PR, O'Hara RB, Simpson GL, Solymos P, Stevens MHH, Wagner H (2016) Vegan: community ecology package. R package version 2:3–3

Ortega-Álvarez R, MacGregor-Fors I (2009) Living in the big city: effects of urban land-use on bird community structure, diversity, and composition. Landscape Urban Plan 3(90):189–195. doi:10.1016/j.landurbplan.2008.11.003

Palmer MW, Pyšek P, Kaplan Z, Richardson DM (2006) Scale dependence of native and alien species richness in north American floras. Preslia 78(4):427–436

Pignatti S (1982) Flora d'Italia. Edagricole, Bologna

Pyšek P (1998) Alien and native species in central European urban floras: a quantitative comparison. J Biogeogr 25(1):155–163. doi:10.1046/j.1365-2699.1998.251177.x

Pyšek P, Jarošik V, Kučera T (2002) Patterns of invasion in temperate nature reserves. Biol Conserv 104:59–70. doi:10.1016/S0006-3207(01)00150-1

Pyšek P, Richardson DM, Rejmánek M, Webster GL, Williamson M, Kirschner J (2004) Alien plants in checklists and floras: towards better communication between taxonomists and ecologists. Taxon 53:131–143

R Core Team (2016) R: a language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria

Rahbek C (2005) The role of spatial scale and the perception of large-scale species-richness patterns. Ecol Lett 8(2):224–239. doi:10.1111/j.1461-0248.2004.00701.x

Rebelo AG, Holmes PM, Dorse C, Wood J (2011) Impacts of urbanization in a biodiversity hotspot: conservation challenges in metropolitan cape town. S Afr J Bot 77:20–35. doi:10.1016/j.sajb.2010.04.006

Ricotta C, Godefroid S, Rocchini D (2010) Patterns of native and exotic species richness in the urban flora of Brussels: rejecting the 'rich get richer' model. Biol Invasions 12:233–240. doi:10.1007/s10530-009-9445-0

Savard JPL, Clergeau P, Mennechez G (2000) Biodiversity concepts and urban ecosystems. Landscape Urban Plan 48(3–4):131–142. doi:10.1016/S0169-2046(00)00037-2

Socolar JB, Gilroy JJ, Kunin WE, Edwards DP (2016) How should Beta-diversity inform biodiversity conservation? Trends Ecol Evol 31:67–80. doi:10.1016/j.tree.2015.11.005

Stohlgren TJ (2007) Measuring plant diversity: lessons from the field. Oxford University Press, Oxford

Sushinsky JR, Rhodes JR, Possingham HP, Gill TK, Fuller RA (2013) How should we grow cities to minimize their biodiversity impacts? Glob Change Biol 19:401–410. doi:10.1111/gcb.12055

Sutherland WJ, Woodroof HJ (2009) The need for environmental horizon scanning. Trends Ecol Evol 24(10):523–527. doi:10.1016/j.tree.2009.04.008

Vähä-Piikkiö I, Kurtto A, Hahkala V (2004) Species number, historical elements and protection of threatened species in the flora of Helsinki, Finland. Landscape Urban Plan 68:357–370. doi:10.1016/S0169-2046(03)00149-X

Vitousek PM, Mooney HA, Lubchenco J, Melillo JM (1997) Human domination of Earth's ecosystems. Science 277(5325):494–499. doi:10.1126/science.277.5325.494

Wiens JA (1989) Spatial scaling in ecology. Funct Ecol 3:385–397