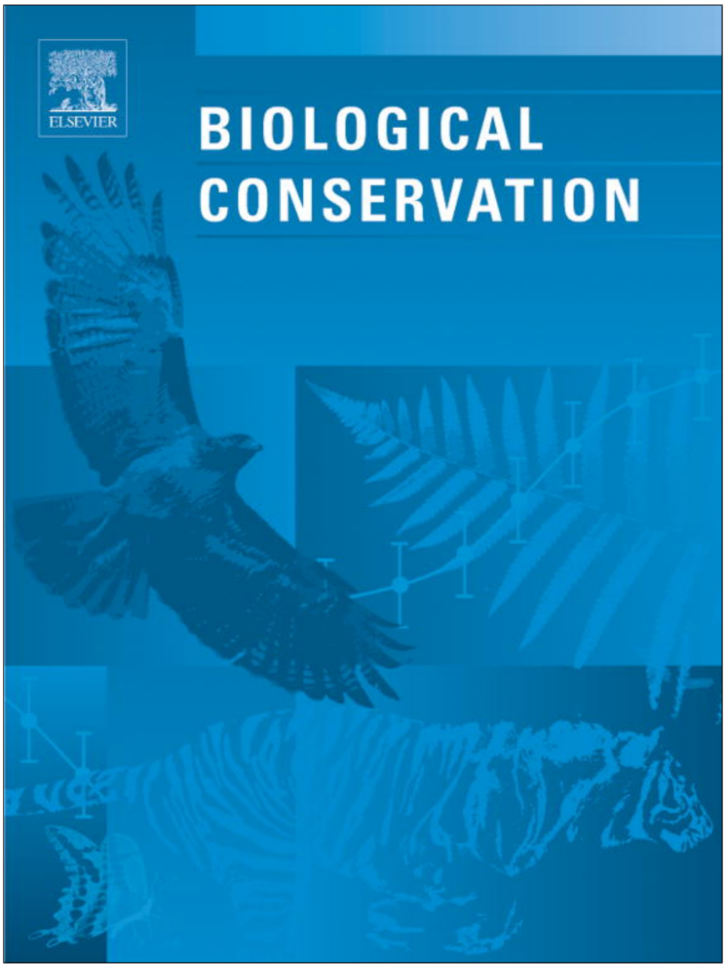
ELSEVIER
BIOLOGICAL
CONSERVATION

Biological Conservation 157 (2013) 285–292

Contents lists available at SciVerse ScienceDirect

Biological Conservation

journal homepage: www.elsevier.com/locate/biocon

Local and management variables outweigh landscape effects in enhancing the diversity of different taxa in a big metropolis

Assaf Shwartz [a,b,*], Audrey Muratet [b], Laurent Simon [a], Romain Julliard [b]

[a] UMR CNRS 7733, Université de Paris 1 Panthéon-Sorbonne, Lab LADYSS, 2 rue Valette, 75005 Paris, France
[b] UMR7204 CNRS-MNHN-UPMC, Lab Conservation des Espèces, Restauration et Suivi des Populations (CERSP), Muséum National d'Histoire Naturelle, CP 51, 55 rue Buffon, 75005 Paris, France

ARTICLE INFO

Article history:
Received 24 April 2012
Received in revised form 3 September 2012
Accepted 14 September 2012

Keywords:
Biodiversity conservation
Diversity-indexes
Landscape-scale
Local-scale
Mulch
Peat
Surrogate
Differential management

ABSTRACT

As the pace of urbanization accelerates, the conservation of urban biodiversity emerges as a rising concern. Urban ecological research has revealed that some green areas in cities can harbor a rich diversity of species that can be enhanced by certain landscape- and local-scale structural planning variables. However, while most studies have been conducted in large greenspaces (e.g., parks, remnants), less effort was made to understand which variables influence biodiversity within small green patches and the efficiency of management practices has been seldom investigated. Here, we explore how management practices interplayed with landscape and structural variables to influence the diversity of plants, birds, butterflies and other pollinating insects in small public gardens (0.5–2.0 ha) in the center of a large metropolis (Paris, France).

Small public gardens hosted significant common biodiversity and the ones that employed a conservation program (i.e., differential management) supported a higher diversity of all taxa and less urbanophile communities of birds and butterflies. Local-scale and management variables were more important in enhancing biodiversity than landscape-scale variables. Specifically, lawns rich in wild plants attracted many pollinators and bird richness increased with tree cover. Pesticides had a negative effect on bird richness, while a higher diversity of habitats and soils (i.e. mulching, peat) increased the diversity of all four taxa. We also found that bird richness could serve as a reasonable surrogate for butterflies and other pollinators. Our results highlight how planning and managing public gardens in the center of a large metropolis can benefit biodiversity, regardless of spatial context.

1. Introduction

Urban development poses some of the greatest threats and challenges for biodiversity conservation in the 21st century (McKinney, 2002). Today, most of the world's population lives in cities (UN, 2008) with an ecological footprint that goes well beyond the boundary of the urban ecosystem (Wu, 2010). Yet it has been shown that cities can harbor a rich diversity of species that sometimes even exceed that found in neighboring greener environments (e.g., Blair, 1996; Rees et al., 2009). This urban nature can also provide a range of socio-economic benefits (TEEB, 2011), including reconnecting people to nature (Miller and Hobbs, 2002). This has stimulated interest in "greening" cities, whether through the growing use of "biodiversity-friendly" management practices, or by extensive work focused on understanding how the spatial arrangement and quality of greenspaces can determine species diversity (reviewed by Sadler et al., 2010).

The potential of small urban greenspaces to improve the functioning of urban ecosystems is relatively unexplored. Indeed, greenspaces in cities are often small, fragmented and isolated, but up until now, most ecological research has focused on large green patches (Matteson and Langellotto, 2010). As a result, very little is known about how these small green islands can contribute to biodiversity conservation and to people. But although small green patches may not provide as many resources or shelter opportunities for different taxa as larger patches, they form interconnected networks that improve the urban matrix permeability (Shanahan et al., 2011). In the context of large urban agglomerations, these small greenspaces also allow people to maintain contact with nature (Miller and Hobbs, 2002), providing ecosystem services that improve city-dweller well-being (Fuller et al., 2007). Thus, understanding how to plan and manage small greenspaces to maintain or even increase biodiversity can be of great value for both city-dwellers and conservationists.

The relative importance of local- vs. landscape-scale variables in influencing urban biodiversity varies among taxa and studied locations (Goddard et al., 2010). For instance, Evans et al. (2009) demonstrated that local variables, such as tree cover, structural

* Corresponding author at: UMR 7204 CERSP, Museum National d'Histoire Naturelle, CP 51, 55 rue Buffon, 75005 Paris, France. Tel.: +33 611396513.
E-mail address: shwar.a@mail.huji.ac.il (A. Shwartz).

0006-3207/$ - see front matter © 2012 Elsevier Ltd. All rights reserved.
http://dx.doi.org/10.1016/j.biocon.2012.09.009

complexity and human disturbance, were more important than landscape variables in determining bird diversity. But most existing avian studies focused on large patches (Goddard et al., 2010). In smaller patches within the urban matrix, Loss et al. (2009) and Shanahan et al. (2011) showed that bird richness was also strongly dependent on landscape factors, such as the heterogeneity of land cover types, distance to natural areas and landscape connectivity. Both landscape- and local-scale variables, such as patch age and diversity of sub-habitats, influence the diversity of vascular plants and pollinators in various urban locations (e.g., Bastin and Thomas, 1999; Brown and Freitas, 2002; Kadlec et al., 2008; Ahrne et al., 2009). But in private gardens, local variables seemed to be more important than landscape ones in determining the diversity of plants and invertebrates (e.g., Thompson et al., 2004; Gaston et al., 2005; Smith et al., 2006). Overall, species diversity does not appear to exhibit similar spatial patterns across different taxa in the urban environment (McDonnell and Hahs, 2008; but see Blair, 1999). But most urban studies focus on a single taxa (McDonnell and Hahs, 2008), and cross-taxonomic studies are needed to better understand the different drivers of urban biodiversity.

While extensive research exists on the relationship between environmental variables and urban biodiversity, the effectiveness of management practices has rarely been studied. The level of management (i.e., grass cutting, pruning and fertilizing) was shown to influence bird diversity in a large urban park (Shwartz et al., 2008). Gaston et al. (2005) demonstrated experimentally that the presence of ponds and nesting-boxes for solitary bees could enhance biodiversity in private gardens, unlike other practices such as leaving deadwood and nettle patches. For pollinators, flower selection and lawn mowing practices have been shown to influence both the richness and the abundance of bees and bumblebees (Ahrne et al., 2009; Kearns and Oliveras, 2009). The diversity and composition of wild plants in urban lawns has been shown to be influenced by mowing practices, public access and the use of pesticides and fertilizers (Kirkpatrick, 2004; Politi-Bertoncini et al., 2012).

As cities become more aware of the multiple benefits provided by nature, they are promoting management practices aimed at increasing biodiversity (TEEB, 2011). One example is the 'differential management' program, which was first developed in Germany during the 1990s as an alternative to intensive horticultural management of urban greenspaces (Aggeri, 2010) and is now widespread in Europe. The program promotes a range of practices for developing sustainable greenspaces in urban areas and one of its objectives is to increase biodiversity (Aggeri, 2010). It therefore recommends some 'biodiversity friendly' practices such as zero pesticides, reuse of organic waste as mulch and the creation of several semi-natural sub-habitats. Although this program has been adopted by several European cities (e.g., Amsterdam, Hamburg, Brussels), we are not aware of any study that has explored the efficiency of these practices in retaining or even increasing urban biodiversity.

This paper investigates how environmental variables interplay with management practices to influence biodiversity in a heavily-developed, densely-populated metropolis (Paris, France). In 2004, the Paris municipality started using the 'differential management' program. It published a set of guidelines for gardeners and managers, who were then able to choose whether to apply all, some or none of them, resulting in a variance in management practices. We therefore used those small public urban gardens as a natural experiment to investigate: (i) which biodiversity they can harbor; (ii) how landscape, local-scale variables and management practices influence the diversity of birds, pollinator insects and wild plants in those public gardens; and (iii) whether some taxa could serve as surrogates for others. Answering those questions could help provide useful guidelines on how to better design and manage small public gardens.

2. Materials and methods

2.1. Study system

The study was carried out in Paris (France), one of the most densely-populated metropolises in Europe. Paris has a green infrastructure consisting of two large parks (~1000 ha) at the periphery of the city, 17 medium-size parks (5–15 ha), over 400 small public gardens (0.1–5.0 ha), and an additional 800 ha of private greenspace (APUR, 2010). Since only public gardens in Paris are gradually applying the differential management program, we conducted our research on 36 small public gardens (0.5–2.0 ha; more details on garden selection in Supplementary material S1).

2.2. Biodiversity surveys

During 2009, we sampled the diversity of three taxa representative of different trophic levels: birds, pollinators and plants. Birds were sampled during the breeding season (April–May) between 30 min before sunrise and 3 h after, using point counts. We visited each garden eight times for 10 min and recorded the species and the number of individuals of every bird seen or heard up to 50 m from the sampling point. Birds flying over the survey area were ignored.

We used two different methods to sample pollinating insects. Diurnal butterflies (*Lepidoptera* sp.) were sampled from June to August on sunny days with a minimum temperature of 18 °C. We used the quadrat method as it was more suitable than a normal transect for sampling the public gardens that were relatively individual- and species-poor. Each garden was visited seven times, by walking in a quadrat of 0.5 ha (without retracing our steps) for 15 min recording any butterfly in sight. All butterflies were identified at the species level, except garden whites, which were grouped at the genus level (i.e., *Pieris*).

To assess the diversity of other pollinators without capturing individuals, we developed a picture-based procedure based on a citizen science protocol used in France (http://www.spipoll.org). We visited each garden seven times for 20 min on sunny days between 9:30 and 17:30. Before the sampling season, we mapped and numbered all the flower patches (i.e., flowerbeds, lawns with flowers, flowering trees and bushes). At each visit, we randomly drew four numbers and sampled the corresponding patches for 5 min by photographing the pollinators on flowers. We later identified the pictured pollinators to morphospecies level (i.e., one species or group of species distinguished from others only by its morphology; Kremen et al., 2011), since 46% of species sampled could only be identified at the species level by capture (e.g., family *Halictidae*).

Finally, we were also keen to explore to what extent small public gardens supported populations of wild plants, beyond the ornamental species that were planted in the gardens but did not form self-replacing populations. The classification of species as wild and cultivated was given by the Conservatoire Botanique National du Bassin Parisien (CBNBP, 2011). Thus, in August we inventoried the presence/absence of wild vascular plant species within the same quadrats as those used for pollinator sampling.

Sampling effort was estimated for birds, butterflies and pollinators in each of the 36 public gardens using sample-based rarefaction curves. We used the observed richness when all public gardens reached accumulation and the average number of species if not (more details in Supplementary material S1).

2.3. Biodiversity indices

Richness of birds, butterflies, plants and pollinator morphospecies (average species per visit), and the average abundance of birds and butterflies were calculated. Indices measuring the affinity of birds, butterflies and plants for urban areas were also estimated. Following Blair (1999), bird species were classified in two groups: urban exploiters (i.e., species that exploit the urban ecosystem; Supplementary material S2) and urban adaptors (i.e., species that adapt to urban environment and live mostly in green areas; Supplementary material S2). We then calculated the urbanophobe bird index as the share of urban adaptors out of the total bird abundance. For butterflies, we calculated the sensitivity of each species to urbanization following Bergerot et al. (2011). The urbanophobe index was calculated as the weighted average of this index based on relative abundance for all public gardens that supported communities of butterflies (i.e., excluding the three gardens with almost no butterflies). We used Biolflor urbanity trait (Kühn et al., 2004) to classify plant species in five categories of urbanity and assess an average plant urbanophobe index per garden. For all three indices, high values reflect urbanophobe communities. We used Pearson correlation coefficients to explore how these indices related to each other.

2.4. Landscape variables

Using ArcMap 9.2, we estimated 14 landscape variables that we expected may influence the diversity of the different taxa in the 36 public gardens. We measured the distance of each garden from the center of Paris (which is negatively correlated with intensity of urbanization; Muratet et al., 2008) and from the nearest large (~1000 ha) urban park (that could act as a source for the nearest gardens). In order to estimate the green cover around each garden, we also used Normalized Difference Vegetation Index (NDVI, IAURIF, 2003) data estimated from satellite imagery (15 m × 15 m resolution data estimated from Landsat 7 Thematic Mapper satellite imagery recorded the 28th August 2000), classified in thirteen classes ranging from zero (concrete) to twelve (dense vegetation). NDVI has been shown to be strongly related to the extent of vegetation cover (Purevdorj et al., 1998). We then calculated the average green proportion for six buffer zones (100–500, 1000 m). Similarly, we calculated the proportion of greenspace (i.e., areas that are defined as green in Land Use Patten map; IAURIF, 2003) around each garden for the same six buffer zones. Since all 14 landscape variables were highly correlated, we used hierarchical partitioning (Mac Nally, 2002) to select the variable with the strongest independent influence on all the biodiversity indices, i.e. the mean NDVI in a 300 m buffer zone around the garden.

2.5. Structural garden variables

Five structural garden variables were digitized using both ArcMap 9.2 and field surveys (garden area, tree cover formed of species with a single trunk and higher than 3 m, bush cover formed of species with several stems and smaller than 3 m, flowerbed cover and lawn cover). We also calculated the Shannon–Wiener index of habitat diversity per garden based on the proportion of cover of each the sub-habitats types (the five sub-habitats listed above, and cover of water, unmanaged areas, and flower meadows – areas seeded with wild flowers and high grasses).

2.6. Management practices

Based on the extent to which public gardens employ the practices recommended under the "differential management" program for the city of Paris, public gardens can obtain 'biodiversity-friendly' certification (notwithstanding the consequences of these practices). We used a student *T*-test to compare the richness of birds, butterflies, wild plants and other pollinators between certified and non-certified public gardens (Fig. 1). However, since the certification involves many different criteria (e.g., water saving, compost), even certified public gardens vary in their management practices.

In order to explore this variance in management practices (i.e., the degree to which different practices are employed in each garden), we interviewed each garden manager with a questionnaire. We assessed five variables that we thought would influence the diversity of species sampled: (1) pesticides – we used a two-level factor indicating the presence/absence of pesticides. Only nine public gardens still use pesticides of the 36 gardens studied; (2) quantity of mulch ranging from 0 (no mulch) to 6 (mulch covers most unpaved parts of the gardens, except lawns). Mulch in Parisian public gardens consists of pellets produced from the organic garden waste; (3) quantity of peat ranging from 0 (no peat) to 5 (covers most unpaved parts of the public gardens excluding lawns); (4) mow height ranging from 4 to 8.5 cm and (5) mowing frequency per month (since public gardens with small lawns were cut in one go). Mowing higher and less frequently was expected to be a biodiversity friendly practice.

2.7. Data analysis

We used nine separate Linear Models to explore the relative influence of management practices, landscape and structural variables on species diversity indices, normal error structures were used when possible. (A scheme showing the statistical design of all analyses used in this paper is presented in Supplementary materials S1.) Since no significant collinearity was found between variables, the landscape variable and all management and structural variables (excluding flowerbed cover for birds, since we did not expect it to influence bird diversity) were entered into the models. We also entered in each model the interactions lawn-cover ∗ mow-height and lawn-cover ∗ mowing frequency, since we expected these practices to be mostly related to lawn cover. All the statistical analyses were done in R.2.12.2 (R Development Core Team, 2011). We first tested for normality assumptions and non-constant error in variance (Breush–Pagan test). Wild plant richness, butterfly and bird abundance were not normally distributed and were modelled in Generalized Linear Models with quasi-Poisson distribution error to account for over-dispersion. We found no spatial auto-correlations among public gardens (Durban–Watson test and Mantel test between taxonomic similarities and geographic distances).

For model selection, we used a model-averaging approach (Burnham and Anderson, 2002; Symonds and Moussalli, 2011).

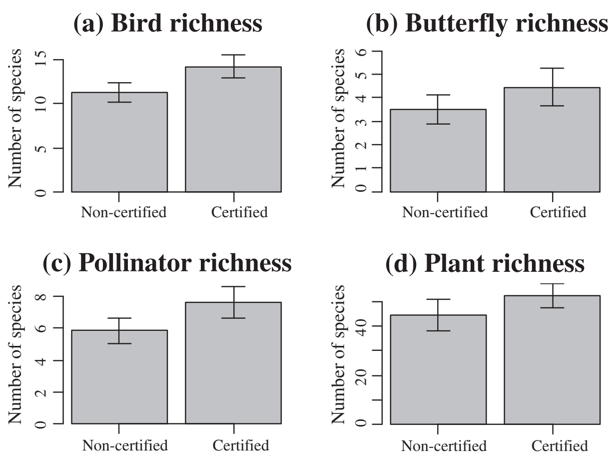

Fig. 1. Differences in richness of birds (a), butterflies (b), pollinators (c) and wild plants (d) between public gardens that were certified as 'biodiversity-friendly' (certified) and public gardens that were not certified (non-certified).

We first ranked all models based on the AICc (corrected Akaike Information Criterion) or QAICc for quasi-Poisson model using the MuMIn package (Barton, 2011). We then considered variables from most parsimonious models (i.e., ΔAICc < 4) by averaging their estimates and standard errors weighted by each model AICc/QAICc (Burnham and Anderson, 2002). Model averaging computed the post-probability (hereafter referred to as *PP*) of an explicative variable influencing the dependant variable taking into account the number of times the term appeared as significant in the selected models. A rule of thumb for using these post-probabilities was to consider that *PP* >0.95, 0.95–0.5, and <0.5 corresponded roughly to the classical *p*-values <0.01, 0.01–0.05, >0.05 (Pellissier et al., 2012). We therefore presented coefficients and standard error for explicative variables that had post-probabilities higher than 0.5 (see Supplementary material S3 for all models and post-probabilities). For each model we also present the adjusted *R*-square, which was calculated as the average *R*-square in the most parsimonious models.

3. Results

Altogether we recorded 30 species of birds (7–20 per garden), 12 species of butterflies (2–9), 74 morphospecies of pollinators (7–36), belonging to 29 different families. We also recorded, 218 wild plants (11–67), of which 19% were naturalized species (see Supplementary material S2 for full species lists).

We found that the 'biodiversity-friendly' public gardens supported a richer biodiversity than non-certified gardens (Fig. 1). Both the richness of birds and the average richness of pollinators were significantly higher in the certified gardens (respectively $t = 3.4$, $p = 0.001$; $t = 2.71$, $p = 0.01$), and the richness of wild plants and butterflies showed a similar but only marginally significant pattern (respectively $t = 1.94$, $p = 0.06$; $t = 1.84$, $p = 0.07$). The biodiversity-friendly public gardens also displayed higher proportions of urbanophobe birds ($t = 2.21$, $p = 0.03$) and tended to support more urbanophobe butterflies than non-certified public gardens.

Several local structural variables and management practices explained the variance of the different diversity indices, whereas the landscape variable influenced only butterfly abundance. Although garden selection and sampling design were made to reduce the effect of garden size on biodiversity, total garden area was positively correlated to the diversity of birds, butterflies and pollinators (Table 1). Among the most parsimonious models, the use of pesticides and mow height had important negative effects on bird richness, while tree cover, mulch and peat exhibited positive effects (Table 1).

Higher use of mulch and peat in public gardens seemed to facilitate conditions for urbanophobe birds. Bush cover and garden size were important in explaining the variance of both richness and abundance of butterflies (Table 1). Lawn cover also had a strong positive effect on butterfly abundance, followed by mulch, the diversity of habitats and a weak effect of NDVI buffer 300. Public gardens with both small lawns and poor tree cover supported more urbanophobe communities of butterflies. Pollinator diversity was positively influenced by the use of peat, and by lawn cover and habitat diversity (Table 1). All parsimonious models for wild-plant richness included a negative interaction between lawn cover and mow height ($PP = 1.0$). In public gardens with small lawn cover, the richness of wild plants increased with mow height, while in public gardens with large lawn cover richness of wild plants decrease with mow height. Wild plant richness was also positively correlated with peat and habitat diversity (Table 1). Urbanophobe wild plant communities were found in public gardens with low tree cover.

The different biodiversity indices were moderately correlated to each other (Table 2), but some interesting patterns emerged. Bird richness was positively correlated to the diversity of butterflies, pollinators, urbanophobe bird index and negatively to the urbanophobe butterfly index (Table 2). Public gardens that had higher abundance of birds supported less urbanophobe communities of birds and wild plants. The richness of pollinators, butterflies, wild plants and the abundance of butterflies were all positively correlated to each other (Table 2). Finally, public gardens that supported urbanophobe butterflies also supported a lower richness of other pollinators and wild plants, and urbanophobe communities of wild plants (Table 2).

4. Discussion

Maintaining biodiversity in urban environments has become an important conservation priority (Jarošik et al., 2011). Our results showed that even small public gardens in the heart of Europe's second largest metropolis (~12 million inhabitants) can host a significant diversity of species from different taxa. Although small public gardens only form a small part of the green infrastructure found in Paris, alongside larger parks, wood remnants, unmanaged areas

Table 1
Estimated average coefficients ±SE for important landscape, structural and management variables (i.e., post-probabilities > 0.5) for the most parsimonious (ΔAICc < 4) generalized linear models ($n = 36$; $^*n = 33$) with normal or quasi-Poisson error selected (butterfly abundance and wild plants richness).

Variable type	Birds richness	Urbanophobe birds	Butterfly richness	Butterfly abundance	Urbanophobe butterfly*	Pollinator richness	Wild plant richness	Urbanophobe plant
Management								
Adjusted R^2	0.72	0.46	0.41	0.59	0.29	0.66	0.57	0.19
Intersect	13.10 ± 0.36	0.21 ± 0.02	4.01 ± 0.23	1.54 ± 0.10	6.98 ± 0.09	6.87 ± 0.24	3.87 ± 0.04	3.06 ± 0.02
Pesticides	−0.76 ± 0.85	–	–	–	–	–	–	–
Mulch	0.52 ± 0.48	0.05 ± 0.02	–	0.16 ± 0.15	–	–	–	–
Peat	0.36 ± 0.40	0.06 ± 0.02	–	–	–	0.96 ± 0.26	0.07 ± 0.04	–
Mow height	−0.42 ± 0.41	–	–	–	–	–	0.002 ± 0.04	–
Mow height * lawn cover	–	–	–	–	–	–	−0.16 ± 0.04	–
Local structural								
Area	2.00 ± 0.38	–	0.72 ± 0.23	0.21 ± 0.10	–	0.62 ± 0.28	–	–
Tree cover	0.90 ± 0.48	–	–	–	−0.23 ± 0.10	–	–	−0.03 ± 0.03
Bush cover	–	–	0.60 ± 0.33	0.19 ± 0.12	–	–	–	–
Lawn cover	–	–	–	0.26 ± 0.09	−0.14 ± 0.11	0.40 ± 0.28	0.04 ± 0.04	–
Diversity of habitats	–	–	–	0.10 ± 0.13	–	0.95 ± 0.26	0.08 ± 0.05	–
Landscape								
NDVI buffer 300	–	–	–	0.08 ± 0.10	–	–	–	–

Table 2
Pearson correlation coefficients for the nine biodiversity indices is presented along with their significance level (*<0.05 and **<0.001).

	Bird richness	Bird abundance	Bird urbanity index	Butterfly richness	Butterfly abundance	Butterfly urbanity index	Pollinator richness	Wild plant richness	Plant urbanity index
Bird richness	1								
Bird abundance	0.03	1							
Bird urbanity index	0.62**	−0.56**	1						
Butterfly richness	0.47**	−0.14	0.44**	1					
Butterfly abundance	0.38*	0.11	0.07	0.56*	1				
Butterfly urbanity index	−0.37*	0.04	−0.29	−0.03	0.11	1			
Pollinator richness	0.53**	0.14	0.17	0.32	0.54**	−0.38*	1		
Wild plant richness	0.17	0.25	0.04	0.12	0.44**	−0.33*	0.45**	1	
Plant urbanity index	−0.26	−0.51**	0.16	0.11	0.13	0.37*	−0.21	−0.07	1

and other private or small green environments, they account for a large part of the city's regional species pool. Birds sampled represented nearly 50% of the birds known to breed in Paris (Malher et al., 2010). Similarly, butterflies sampled accounted for over half of the total species sampled in 135 sites in the Paris region (Ile-de-France; Bergerot et al., 2011), while other pollinators accounted for 44% of morphospecies reported in a citizen science project of 406 sites in the Paris region (Deguines et al., 2012). The plant species observed in this study corresponded to over 30% of the flora observed in almost one thousand sites of an urban department in the same region (Muratet et al., 2008). However, the vast majority of species sampled in the small public gardens were common species in the Paris region. Common species are highly important for conservation, since they contribute much of the structure, biomass and energy turnover of an ecosystem (Gaston, 2010). They are also frequent victims of habitat loss and species invasions, which could have a profound influence on ecosystems and the services they provide. In the urban context, these common species form interactions with people, which could influence their well-being and shape their relation with nature (Miller and Hobbs, 2002).

Developing sustainable cities is one of the great challenges for urban planners, local authorities and conservationists (Wu, 2010). Programs like the 'differential management' program that aim to find a subtle balance between horticultural traditions and 'natural' management could contribute to these efforts. However, the evaluation of management programs is crucial to understanding their value for conservation (Ferraro and Pattanayak, 2006). The differential management program promotes the creation of semi-natural habitats and the use of 'environmental friendly' practices in greenspaces (Aggeri, 2010). One of its objectives is to safeguard high biodiversity. In our study, the 20 'biodiversity-friendly' public gardens supported a richer diversity of species than the 16 public gardens without this label. On this scale, it seems that the differential management program benefits various taxa. However, it is important to further explore the relationship between each taxon, management practices and garden characteristics if we are to validate those practices to promote sustainable conservation.

4.1. Landscape, structural variables and species diversity

In the context of a large metropolis, we found that local-scale variables and management practices had more influence on species diversity in small public gardens than landscape-scale effects. The green cover index only weakly influenced the abundance of butterflies. This is likely explained by the movement of migratory butterflies along green corridors or stepping stones (Baum et al., 2004). Indeed, the migrating painted lady (*Cynthia cardui*) accounted for 61% of total butterfly abundance in our survey, and when we rerun the model excluding migrating species, the green cover index no longer had a significant effect on butterfly abundance (*PP* = 0.16). The absence of landscape effect for birds appears to be consistent with other studies focused mostly on large greenspaces (Evans et al., 2009), but is less consistent with recent work that investigated the effect of landscape variables on small green patches in large cities (Shanahan et al., 2011). Our results imply that adequate planning and management of small public gardens could suffice to increase species diversity (of some taxa) irrespective of the green context these gardens are located in. While large greenspaces can support more species than small patches, budgetary and spatial constraints often prevent conservation at such large scales (Loss et al., 2009). Therefore, in such cases (but not only those) small gardens (public and private) provide an excellent opportunity to increase the quality of biodiversity in the city.

Many studies demonstrate positive relationships between area, vegetation diversity and species diversity for various taxa (Goddard et al., 2010). Although we only considered a small range of garden sizes (0.5–2 ha), area was important in four out of the nine indices studied, thus even small changes in garden area may increase species diversity. Similarly, the positive relationship between vegetation diversity and biodiversity is well-established in cities (e.g., Kearns and Oliveras, 2009; Jarošik et al., 2011), especially between the diversity of woody species and birds (Evans et al., 2009). Accordingly, we found that public gardens with a large proportion of tree cover supported more bird species, but also a lower proportion of urbanophobe plants and butterflies. Trees are essential for protection, nesting and feeding of many urban breeding birds (Fontana et al., 2011). However in our public gardens, trees were generally used to shade pathways and were thus associated to concrete or compacted soil cover, where urbanophobic plants could hardly survive. For example, *Sagina procumbens*, a plant very tolerant to trampling and considered as urbanophile (urbanity index = 2), was only found in public gardens with high tree cover (>40%). Since the most urbanophile butterfly species were woodland species such as the speckled wood (*Pararge aegeria*) and comma (*Polygonia c-album*), high tree cover was associated with urbanophobic communities of butterflies.

In cities, the richness of pollinators has been found to be positively correlated to the diversity of nectar-giving flowers (Matteson et al., 2008; Ahrne et al., 2009; Kearns and Oliveras, 2009). In this study, flowerbed cover was not correlated to pollinator richness, but lawn cover was found to be positively correlated with butterfly abundance and with the richness of other pollinators and wild plants. For instance, a 10% increase in lawn cover could add four pollinator species. Flowerbeds contained mostly flowers selected for the production of numerous petals to the detriment of nectar production (Comba et al., 1999). Overall, 87% of wild plants inventoried were either nectar or pollen resources for pollinators (e.g. *Cirsium arvense*, *Cirsium vulgare*, *Trifolium repens*). Improving lawn quality was one of the priorities fixed by the municipality of Paris as part of the 'differential management' program, as commonly 'municipal lawns' are not pollinator friendly. This involved practices facilitating conditions for wild plants such as aeration, less weeding, leaving grass longer when mowing and reducing mowing frequency. These initiatives improved the ability of lawns to support wild flowers essentially found in lawns (69% of sampled wild plants) and enhance the diversity of pollinators, which were found to be correlated to wild plant richness. Thus, lawn management in Parisian public gardens represent a unique and successful case study, in which conservation campaign has facilitated the conditions for biodiversity in lawns. Additional conservation campaigns, that encourage gardeners to design pollinator-friendly flowerbeds, could help in further enhancing pollinator's diversity in small public gardens.

Butterfly richness and abundance were positively correlated to bush cover. This could be explained by the over-representation of the exotic butterfly bush, *Buddleja davidii*, considered as invasive in the region and known to attract several butterflies and other pollinators (Tallent-Halsell and Watt, 2009). A comparison between public gardens with ($n = 21$) and without ($n = 15$) butterfly bush revealed that butterfly abundance was almost three times higher (3.27 vs. 9.54; $t = 2.95$, $p = 0.007$) and butterfly richness was also higher (2.8 vs. 4.8; $t = 2.71$, $p = 0.01$) in gardens with the butterfly bush. This result raises an interesting debate regarding the use of exotic plants in urban conservation management (Prévot-Julliard et al., 2011), since besides its positive effect on butterflies, this species could have detrimental effects on the native flora (Tallent-Halsell and Watt, 2009). However, a cultivated sterile hybrid highly attractive to butterflies has been developed, which offers an opportunity to plant this species safely in urban greenspaces.

4.2. The differential management program and species diversity

The main objective of this study was to understand how different practices proposed by the 'differential management' programs can facilitate the conservation of biodiversity in small public gardens. This program aims to improve the quality of garden habitats, also by promoting structural changes, such as introducing less-managed sub-habitats such as ponds, meadows and unmanaged areas (Aggeri, 2010). Indeed, public gardens with higher habitat diversity supported a higher richness of wild plants and pollinators, and more 'natural' communities of those taxa. In the context of small public gardens, the scale of change was minor (i.e., introducing sub-habitat in the scale of 30–50 m^2) and those changes seemed to be less important for birds, yet significant for plants and pollinators. Increasing the diversity of habitats can improve natural processes, such as recruitment and germination, and also provide resources for various garden pollinators, especially for less mobile species such as bees and bumblebees (Matteson and Langellotto, 2010).

The differential management program also promotes a range of soil management practices, of which use of mulch and peat and removal of pesticides may be the most prevalent. We found that pesticides only had a negative influence on bird richness. The effect of pesticides has rarely been studied in urban environments, but some evidence points to a negative effect of certain chemical inputs on plant richness, rarity and bird diversity (Geiger et al., 2010; Politi-Bertoncini et al., 2012). The impact of pesticides depends on their types (insecticides, herbicides, fungicides, etc.) and the quantity applied. Since these data were not obtained for the nine public gardens that still use pesticides in Paris, a more thorough approach is required to better understand how pesticides influence biodiversity. In contrast, mulch is an organic cover placed over the soil that retains moisture and provides nutrients that stimulate soil activity, resulting in improved soil fertility (Hanula and Horn, 2011). Ground foragers such as blackbirds, song thrushes and robins may especially profit from mulching. Indeed, their abundance was positively correlated with mulch (Spearman's rho = 0.36 $p = 0.03$). The abundance of butterflies appeared to benefit from the use of mulch. This might be because conditions for wild plants were slightly improved. Indeed, mulch appeared significant in about 40% of most parsimonious models for wild plant richness ($PP = 0.40$). Furthermore, mulch could also improve conditions that influence plant abundance and the quality of their flower production, which could explain the positive relation between mulch and butterfly's abundance. Peat is recommended for its ability to acidify the soil and favor particular cultivated or wild plant communities adapted to this soil acidity. Its use effectively creates an additional ecosystem in the garden and could thus be associated with an increase in the number of species observed. Indeed, we found that peat use had a strong positive effect on pollinator and wild plant richness. Peat areas were also mostly fenced-off, which can facilitate conditions for urbanophobic birds such as bush nesting birds (e.g., song thrush, European robin). Indeed, bush nester abundance was positively correlated with peat (Spearman's rho = 0.62, $p < 0.001$). However, this practice is not promoted by the municipality of Paris, since its mining damages natural environments.

While mowing frequency did not influence the diversity indices, lawn area and the mow height had a significant influence on wild plant richness. In private gardens, mowing was frequently also found to have little effect on plant diversity (Thompson et al., 2004). However, wild plant richness was found to increase more rapidly in private gardens with small lawns than in gardens with larger ones (Thompson et al., 2004). Similarly, we found that public gardens with a small lawn cover were poorer in wild plants than public gardens with a larger lawn cover (44.6 ± 3.0 species vs. 51.7 ± 2.7 mean ± SE). Mowing lawns short creates a disturbance that can increase wild plant diversity by facilitating the conditions for stress-tolerant species and ruderal species that are able to colonize these pioneer habitats. By contrast, high lawns are shaded, essentially by species with high competitive power (e.g., *Lolium perenne*), thus offering fewer available niches and supporting a smaller diversity of wild plants. In Paris, leaving grass longer is considered a 'biodiversity-friendly' practice for plants and pollinators, as this is what is found in more 'natural environments' (Kearns and Oliveras, 2009). Overall, lawn quality in the public gardens was high, supporting a rich diversity of wild plants. When considering the height of mowing, we found that wild plant richness could benefit from a mixed strategy based on the size of lawns and on the plants targeted for conservation (e.g., ruderal or competitive species). However, the range of lawn heights studied here remained very small (4–8.5 cm) and did not allow us to compare the diversity of lawns vs. meadows, which are known for their high floristic diversity (Muratet et al., 2008).

4.3. Biodiversity indices

Although the use of a single indicator taxa or biodiversity index has been criticized for conservation studies (Caro and O'Doherty, 1999), much of the conservation and ecological research and policy still leans on surrogacy (Rodrigues and Brooks, 2007). In the urban context, some studies have shown that different taxa respond similarly to urbanization, especially along the urban gradient (e.g., Blair, 1999; Jarošik et al., 2011). Here, we found that most biodiversity indices were only moderately correlated to each other. This demonstrates the importance of monitoring several taxa if we are to understand how to conserve biodiversity in the urban environment (Jarošik et al., 2011). However, we did find some correlations among the nine indicators, with mostly bird richness having a weak surrogacy power over the other taxa. Similarly to Blair (1999), we found that bird richness was positively correlated to butterfly richness and abundance, urbanophobe bird index and pollinator richness. On the other hand, bird abundance, which was mostly driven by urban exploiters (75.7% of individuals observed), can serve as an indicator for species-poor public gardens, as it was negatively correlated with the bird and plant urbanophobe indices. Although this study was conducted within the urban ecosystem, these findings coincide with landscape-scale studies along the urban–rural gradient that showed contradictory responses of bird richness and abundance to urbanization (e.g., Blair, 1996). Birds, which are particularly well studied in cities and located high in the food chain (Evans et al., 2009), could therefore serve as reasonable surrogates for pollinators and together with wild plants could serve as useful indices for monitoring biodiversity patterns in small public gardens.

5. Conclusion

Although small public gardens may not harbor as much diversity as large green areas in the urban environment, it has become clearer that small patches (i.e., private and public gardens) can contribute to general urban conservation efforts (Goddard et al., 2010). Our results coincide with other studies (e.g., Loss et al., 2009; Fontana et al., 2011) showing that even these habitats can support a significant level of common biodiversity, even in the heart of a large metropolis. Moreover, since many city-dwellers frequently visit those gardens in their daily life, these gardens may play an important role in reconnecting people with nearby nature (Miller and Hobbs, 2002).

Our results further underline that the 'differential management' program could be useful in increasing biodiversity of small public gardens. Improving the quality of public gardens (i.e., mulch, peat, zero pesticide and mowing practices) and introducing a diversity of sub-habitats (ponds, flower meadows, unmanaged patches) can have an important influence on biodiversity, regardless of the landscape context in which the public gardens are located. Such practices are often easier and faster to apply in comparison with efforts to increase the green index of a city or to manage large parks (Loss et al., 2009). Nevertheless, our results also demonstrate the importance of setting specific goals for conservation programs and of validating those goals (Ferraro and Pattanayak, 2006), since the efficiency of some practices could vary among taxa and locations. Therefore, studying several taxa and locations is crucial if we are to understand urban biodiversity (Jarošik et al., 2011). Research on the role of management practices on urban biodiversity is still needed and those findings may help to pinpoint some practices that can improve the quality of the urban ecosystem for a range of taxa and locations. City-planners and decision-makers could then use this information to prioritise conservation efforts and (i.e., select practices that answer their conservation program goals).

Acknowledgements

We would like to thank the Paris municipality's Green Areas and Environment Department (DEVE) for their help in conducting this research and two anonymous reviewers for their helpful comments of an earlier version of this manuscript. This study was supported by the Ile-de-France Sustainable Development Research Network (R2DS Ile-de-France).

Appendix A. Supplementary material

Supplementary data associated with this article can be found, in the online version, at http://dx.doi.org/10.1016/j.biocon.2012.09.009.

References

Aggeri, G., 2010. Inventer les Villes-natures de Demain, Gestion Différenciée Durable des Espaces Verts. Éducagri éd. Dijon.

Ahrne, K., Bengtsson, J., Elmqvist, T., 2009. Bumble bees (*Bombus* spp) along a gradient of increasing urbanization. PLoS ONE 4, e5574.

APUR (Atelier Parisien d'Urbanisation), 2010. Situation et Perspective de la Place de la Nature à Paris. <http://www.apur.org/> (accessed March 2012).

Barton, K., 2011. MuMIn: Multi-model Inference. R Package Version 1.0.0. <http://ww.CRAN.Rproject.org/package=MuMIn>.

Bastin, L., Thomas, C.D., 1999. The distribution of plant species in urban vegetation fragments. Landscape Ecol. 14, 493–507.

Baum, K.A., Haynes, K.J., Dillemuth, F.P., Cronin, J.T., 2004. The matrix enhances the effectiveness of corridors and stepping stones. Ecology 85, 2671–2676.

Bergerot, B., Fontaine, B., Julliard, R., Baguette, M., 2011. Landscape variables impact the structure and composition of butterfly assemblages along an urbanization gradient. Landscape Ecol. 26, 83–94.

Blair, R.B., 1996. Land use and avian species diversity along an urban gradient. Ecol. Appl. 6, 506–519.

Blair, R.B., 1999. Birds and butterflies along an urban gradient: surrogate taxa for assessing biodiversity? Ecol. Appl. 9, 164–170.

Brown Jr., K.S., Freitas, A.V.L., 2002. Butterfly communities of urban forest fragments in Campinas, Sao Paulo, Brazil: structure, instability, environmental correlates, and conservation. J. Insect Conserv. 6, 217–231.

Burnham, K.P., Anderson, D.R., 2002. Model Selection and Multimodel Inference. A Practical Information-theoretic Approach, third ed. Springer, New-York.

Caro, T.M., O'Doherty, G., 1999. On the use of surrogate species in conservation biology. Conserv. Biol. 13, 805–814.

CBNBP, 2011. Conservatoire Botanique National du Bassin Parisien. <http://cbnbp.mnhn.fr/cbnbp/> (in French).

Comba, L., Corbet, S.A., Hunt, L., Warren, B., 1999. Flowers, nectar and insect visits: evaluating British plant species for pollinator-friendly gardens. Ann. Bot. 83, 369–383.

Deguines, N., Julliard, R., de Flores, M., Fontaine, C., 2012. The whereabouts of 710 flower visitors: contrasting land-use preferences revealed by a country-wide 711 survey based on citizen science. PLoS One 7, e45822.

Evans, K.L., Newson, S.E., Gaston, K.J., 2009. Habitat influences on urban avian assemblages. Ibis 151, 19–39.

Ferraro, P.J., Pattanayak, S.K., 2006. Money for nothing? A call for empirical evaluation of biodiversity conservation investments. PLoS Biol. 4, 482–488.

Fontana, S., Sattler, T., Bontadina, F., Moretti, M., 2011. How to manage the urban green to improve bird diversity and community structure. Landscape Urban Plann. 101, 278–285.

Fuller, R.A., Irvine, K.N., Devine-Wright, P., Warren, P.H., Gaston, K.J., 2007. Psychological benefits of greenspace increase with biodiversity. Biol. Lett. 3, 390–394.

Gaston, K.J., 2010. Valuing common species. Science 327, 154–155.

Gaston, K.J., Smith, R.M., Thompson, K., Warren, P.H., 2005. Urban domestic gardens (II): experimental tests of methods for increasing biodiversity. Biodivers. Conserv. 14, 395–413.

Geiger, F., Bengtsson, J., Berendse, F., et al., 2010. Persistent negative effects of pesticides on biodiversity and biological control potential on European farmland. Basic Appl. Ecol. 11, 97–105.

Goddard, M.A., Dougill, A.J., Benton, T.G., 2010. Scaling up from gardens: biodiversity conservation in urban environments. Trends Ecol. Evol. 25, 90–98.

Hanula, J.L., Horn, S., 2011. Removing an exotic shrub from riparian forests increases butterfly abundance and diversity. For. Ecol. Manage. 262, 674–680.

IAURIF, 2003. Institute for Planning and Development of the Paris Ile-de-France Region. <http://www.iaurif.org> (accessed May 2011).

Jarošik, V., Konvicka, M., Pysek, P., Kadlec, T., Benes, J., 2011. Conservation in a city: do the same principles apply to different taxa? Biol. Conserv. 144, 490–499.

Kadlec, T., Benes, J., Jarosik, V., Konvicka, M., 2008. Revisiting urban refuges: changes of butterfly and burnet fauna in Prague reserves over three decades. Landscape Urban Plann. 85, 1–11.

Kearns, C.A., Oliveras, D.M., 2009. Environmental factors affecting bee diversity in urban and remote grassland plots in Boulder, Colorado. J. Insect Conserv. 13, 655–665.

Kirkpatrick, J.B., 2004. Vegetation change in an urban grassy woodland 1974–2000. Aust. J. Bot. 52, 597–608.

Kremen, C., Ullmann, K.S., Thorp, R.W., 2011. Evaluating the quality of citizen-scientist data on pollinator communities. Conserv. Biol. 25, 607–617.

Kühn, I., Durka, W., Klotz, S., 2004. BiolFlor – a new plant-trait database as a tool for plant invasion ecology. Divers. Distrib. 10, 363–365.

Loss, S.R., Ruiz, M.O., Brawn, J.D., 2009. Relationships between avian diversity, neighborhood age, income, and environmental characteristics of an urban landscape. Biol. Conserv. 142, 2578–2585.

Mac Nally, R., 2002. Multiple regression and inference in ecology and conservation biology: further comments on identifying important predictor variables. Biodivers. Conserv. 11, 1397–1401.

Malher, F., Lesaffre, G., Zucca, M., Coatmeur, J., 2010. Oiseaux Nicheurs de Paris: Un Atlas Urbain. Delachaux et Niestle, Paris.

Matteson, K.C., Langellotto, G.A., 2010. Determinates of inner city butterfly and bee species richness. Urban Ecosyst. 13, 333–347.

Matteson, K.C., Ascher, J.S., Langellotto, G.A., 2008. Bee richness and abundance in New York City urban gardens. Ann. Entomol. Soc. Am. 101, 140–150.

McDonnell, M.J., Hahs, A.K., 2008. The use of gradient analysis studies in advancing our understanding of the ecology of urbanizing landscapes: current status and future directions. Landscape Ecology 23, 1143–1155.

McKinney, M.L., 2002. Urbanization, biodiversity, and conservation. Bioscience 52, 883–890.

Miller, J.R., Hobbs, R.J., 2002. Conservation where people live and work. Conserv. Biol. 16, 330–337.

Muratet, A., Porcher, E., Devictor, V., Arnal, G., Moret, J., Wright, S., Machon, N., 2008. Evaluation of floristic diversity in urban areas as a basis for habitat management. Appl. Veg. Sci. 11, 451–460.

Pellissier, V., Cohen, M., Boulay, A., Clergeau, P., 2012. Birds are also sensitive to landscape composition and configuration within the city centre. Landscape Urban Plann. 104, 181–188.

Politi-Bertoncini, A., Machon, N., Pavoine, S., Muratet, A., 2012. Effects of management on urban lawn floristic communities. Landscape Urban Plann. 105, 53–61.

Prévot-Julliard, A.C., Clavel, J., Teillac-Deschamps, P., Julliard, R., 2011. The need for flexibility in conservation practices: exotic species as an example. Environ. Manage. 48, 882–884.

Purevdorj, T., Tateishi, R., Ishiyama, T., Honda, Y., 1998. Relationships between percent vegetation cover and vegetation indices. Int. J. Remote Sens. 19, 3519–3535.

Rees, M., Roe, J.H., Georges, A., 2009. Life in the suburbs: behaviour and survival of a freshwater turtle in response to drought and urbanization. Biol. Conserv. 142, 3172–3181.

Rodrigues, A.S.L., Brooks, T.M., 2007. Shortcuts for biodiversity conservation planning: the effectiveness of surrogates. Ann. Rev. Ecol. Evol. Syst. 1, 713–737.

Sadler, J., Bates, A., Hale, J., James, P., 2010. Bringing cities alive: the importance of urban green spaces for people and biodiversity. In: Gaston, K.J. (Ed.), Urban Ecology. Cambridge University Press, Cambridge, pp. 172–201.

Shanahan, D.F., Miller, C., Possingham, H.P., Fuller, R.A., 2011. The influence of patch area and connectivity on avian communities in urban revegetation. Biol. Conserv. 144, 722–729.

Shwartz, A., Shirley, S., Kark, S., 2008. How do habitat variability and management regime shape the spatial heterogeneity of birds within a large Mediterranean urban park? Landscape Urban Plann. 84, 219–229.

Smith, R.M., Warren, P.H., Thompson, K., Gaston, K.J., 2006. Urban domestic gardens (VI): environmental correlates of invertebrate species richness. Biodivers. Conserv. 15, 2415–2438.

Symonds, M.R.E., Moussalli, A., 2011. A brief guide to model selection, multimodel inference and model averaging in behavioural ecology using Akaike's information criterion. Behav. Ecol. Sociobiol. 35, 13–21.

Tallent-Halsell, N.G., Watt, M.S., 2009. The invasive *Buddleja davidii* (butterfly bush). Bot. Rev. 75, 292–325.

TEEB (The Economics of Ecosystems and Biodiversity), 2011. TEEB Manual for Cities: Ecosystem Services in Urban Management. <http://www.teebweb.org/> (accessed July 2011).

Thompson, K., Hodgson, J.G., Smith, R.M., Warren, P.H., Gaston, K.J., 2004. Urban domestic gardens (III): composition and diversity of lawn floras. J. Veg. Sci. 15, 373–378.

UN (United Nations), 2008. World Urbanization Prospects: The 2007 Revision Population Database. <http://www.esa.un.org/unup/> (accessed March 2012).

Wu, J., 2010. Urban sustainability: an inevitable goal of landscape research. Landscape Ecol. 25, 1–4.