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Identifying indicator plant species of habitat quality and invasibility as a guide for peri-urban forest management

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Abstract. A floristic survey has been carried out in a peri-urban forest, the Sonian Forest in Brussels, to identify indicator plant species in the herbaceous layer, which could be used as an aid within the framework of a more sustainable management of the forest. Three hundred and seventy two (372) taxa have been identified, 33 of which are non-native (i.e. non-indigenous species regarding the study area, whether invasive or not). Criteria of habitat quality that have been chosen are the species richness, the commonness of the habitat, based on constitutive species, and its invasibility (vulnerability for invasion). On the basis of a comparison of the value of these criteria when each considered (potential indicator) species is present or not, 17 species have been recognised as reliable indicators of at least one of these three criteria. In particular, vegetation types containing either *Anthriscus sylvestris*, *Galeopsis tetrahit* or *Senecio ovatus* were found to be more susceptible to invasion than other habitats. The way to how the predictability of invasions might be effectively used as a management tool is discussed. Furthermore, we found a positive significant correlation (Bonferroni corrected probabilities) between the species richness and luminosity factor (derived from Ellenberg's indices), and the proportion of grassland and wetland species. The species richness was significantly negatively correlated with the proportion of woodland species. An increase in commonness was significantly correlated with a decrease in the proportion of geophytes. The usefulness of these results as an ecological basis for forest management is discussed.

Introduction

Forested residual spaces are unique resources that contribute to the diversity of the urban ecosystem. The planning and management of forested areas in the urban environment have become increasingly important as the focus of urban forestry has shifted from individual tree planting and maintenance to the role that urban forests play in the larger urban ecosystem (Airola and Buchholz 1984). As a valuable natural resource, urban forests may provide a number of primary and secondary benefits, including climatic regulation, air pollution and noise reduction, watershed protection, recreational opportunities, outdoor education, wood products, habitat resources for wildlife and flora, and aesthetics (Doolittle 1969). Nevertheless, for any urban forest, especially for those which are exploited, the composition and distribution of the herb layer constitutes a largely undervalued resource. Yet sufficient evidence exists that, in a forest ecosystem, herbaceous vegetation is far more sensitive than woody strata and indicates more precisely variations of soil conditions (Tanghe and Froment 1968). This is even more pronounced in areas with intensively managed forests.

For site-specific monitoring, indicators can be identified for use in assessing environmental quality (Spellerberg 1994). There are two very different concepts of indicator species. One is a species the presence or absence of which indicates some environmental condition: metals, pesticides, acidity, overclearing, compaction, temperature-shock, winterfrosts, etc. There are plenty indicators of that type, both positive and negative. In order to judge the success or failure of management regimes, we also need to identify indicators of the conservation value of forests. In this framework, studies are required to test relationships between the presence of potential indicator species and criteria of habitat quality. This introduces the second concept of indicator species, which is a species indicating 'biodiversity' *per se*: i.e., the presence of this species is correlated with the presence of (1) lots of other, unrelated species, or (2) other species somehow judged to be valuable, or endangered, or otherwise in need of some management. We use this latter definition of indicator plants in our study. Because the relationships between potential indicator species and biodiversity are not well established (Lindenmayer et al. 2000) and the identification of indicators has only rarely involved statistical tests (Mark and Lawesson 2000), we suggested an approach which relates the presence of particular species to biodiversity, using statistical methods.

Our study brings an ecological perspective to the management of a peri-urban forest, the Sonian Forest in Brussels. Our goal was to assess quantitatively the quality of the herbaceous vegetation of this forest and to determine the possibility of identifying some good indicators of ecosystem attributes, such as species richness, commonness and invasibility (vulnerability to invasion), based on the distribution of the flora on a grid-map throughout the forest.

Study site

The research was conducted in the Sonian Forest, which lies south of Brussels (50°47′ N; 4°26′ E). It is a remnant of the huge forest that covered the whole of Western Europe after the last Ice Age. The forest covers an area of 4383 ha, 1654 ha of which are situated within the administrative limits of the Brussels Capital Region. The present study deals with that part of the forest included within the boundaries of the city (Figure 1). One million inhabitants live in Brussels, which covers an area of 16000 ha and 10% of the Belgian population live within 10 km from the Sonian Forest margin. Previous studies have described in detail the flora of the city (Godefroid 1996a) and the forest (Godefroid 1995).

Some 20000 years ago, sandstone and flintstone formed the upper layer in the area of the Sonian Forest. After the last Ice Age, this layer was covered with loess. Today, almost the whole surface of the forest (95%) is composed of a 3–4 m thick silt layer, which corresponds to the loess deposition. The forest ranges in altitude from 65 to 130 m a.s.l. The climate of the area is temperate and humid, with a growing season of 7 months (April–October). Mean annual temperature is 9.9 °C,

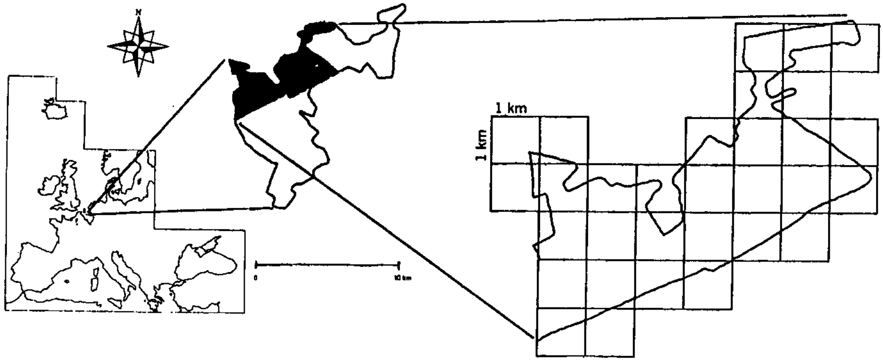

Figure 1. Location of the study area (in black) compared with the total extent of the Sonian Forest (in white) and a systematic grid of 1 km^2 cells covering the study area.

annual precipitation is 835 mm. Originally, the Sonian Forest was an oak–hornbeam forest (dominated by *Quercus robur* and *Carpinus betulus*). Since the plantation work of the Austrian administration at the end of the 18th century, it is now composed of 85% of European beech (*Fagus sylvatica*). Except beech, few other woody species are found. Seven percent of the forest surface is occupied by oak stands (*Quercus robur*) and 8% is represented by introduced conifer stands (*Pinus sylvatica, Larix decidua, Picea abies*). Many paths and roads cross the forest. One counts currently 17 km of roads open to motor vehicles, 152 km of roads closed to motor vehicles, 84 km of footpaths, 30 km of bicycle paths and 50 km of bridle paths for an area of 1654 ha.

Methods

Data collection

We used a systematic grid covering the forest from its edges to the interior (Figure 1). The study area includes 31 1 km^2 cells which are totally (interior-cells) or partially (edge-cells) covered by the forest. Of these 31 grid-cells, six edge-cells have been discarded because they contained too few forested habitat or included some man-made infrastructure inside the forest (e.g. racecourse). Within each of the 25 remaining cells, all vascular species were recorded wherever they occurred within the forest. In order to avoid undersampling because of the seasonal variation, each 1 km^2 cell was surveyed two times along the growing season (early spring, summer or early autumn). On each visit, the duration of search was long enough to reach the point where it was difficult to add further species to the list. Field work was performed during three growing seasons, from 1992 to 1994.

Data analyses

Three criteria of habitat quality have been chosen: species richness, commonness and invasibility. Within each grid-cell, values for these criteria have been calculated. Species richness is defined as the total number of vascular plants in each cell within the grid. Commonness is calculated based on the average of the rarity index of all species present in the grid-cell. This species rarity index is given for north Belgium by the arithmetical scale of Stieperaere and Fransen (1982). This scale classifies each species of the Belgian flora within one of the 10 rarity levels, corresponding to a frequency class of the Atlas of the Belgian and Luxemburger flora (van Rompaey and Delvosalle 1979): rarity level 1: frequency class 0–10%; 2: 11–20%; 3: 21–30%; 4: 31–40%; 5: 41–50%; 6: 51–60%; 7: 61–70%; 8: 71–80%; 9: 81–90%; 10: 91–100%. According to these authors, the rarity index 1 qualifies a species which is very rare, while the rarity index 10 refers to a very common species. For this reason and for a better understanding, we preferred to use the expression 'commonness', which is here defined as the opposite of rarity. Invasibility is referred to as the susceptibility of an ecosystem to invasion by non-native species. Within the framework of this study, this parameter was quantified by the proportion of non-native species found within each grid-cell. Non-native species are defined as non-indigenous plants in the study area, according to Lambinon et al. (1992).

Besides species richness, commonness and invasibility, we used environmental parameters and species ecological groups to characterise habitats within each 1 km^2 cell. A quantitative estimation of ecological demands of each species is given by the well-known and validated indicator values of Ellenberg et al. (1991). These values were used to calculate an average for each grid-cell, based on constitutive species, expressing soil nitrogen (N), soil reaction (R), soil humidity (F) and site luminosity (L).

Nomenclature and species life forms are given by Lambinon et al. (1992). Species ecological groups are determined according to Stieperaere and Fransen (1982). As defined by these authors, different units recognised in this study are:

1. group 1: pioneers from disturbed, artificial, anthropised habitats: road verges, dry waste lands, fields.
2. group 2: pioneers from disturbed semi-natural habitats, on open humid soils.
3. group 4: river bank and fresh-water plants.
4. group 5: plants from mesotrophic to eutrophic humid to damp grasslands.
5. group 6: plants from dry grasslands, walls and rocks.
6. group 7: plants from heathlands, peat-bogs and alkaline marshlands.
7. group 8: species from clear-felled areas.
8. group 9: forest plants.

The proportion of species belonging to each of these groups has been calculated for each grid-cell. The path and road length in each cell is calculated with the help of a curvimeter on topographic maps of the National Geographic Institute (scale: 1:10,000).

We used Pearson's correlation coefficient to examine whether species richness,

Table 1. Relative occurrence of non-native species in the Sonian Forest (Brussels).

Non-native species	L	F	R	N	LF	Ecol. gr.	Ra	Freq. (%)
Impatiens parviflora	4	5	–	6	t	8	1	92
Castanea sativa	5	–	4	–	p	9	5	64
Quercus rubra	–	–	–	–	p	–	–	48
Fallopia japonica	8	8	5	7	g	1	3	44
Alnus incana	6	7	8	–	p	9	3	40
Robinia pseudoacacia	5	4	–	8	p	9	7	40
Erigeron canadensis	8	4	–	5	t	1	8	36
Aesculus hippocastanum	–	–	–	–	p	–	–	36
Prunus serotina	6	5	–	–	p	9	4	32
Solidago gigantea	8	6	–	7	h	1	1	32
Galinsoga ciliata	7	4	6	7	t	1	3	20
Symphoricarpus albus	–	–	–	–	p	–	–	20
Populus alba	–	–	–	–	p	–	–	16
Mespilus germanica	6	4	6	–	p	8	1	12
Solidago canadensis	8	–	–	6	h	1	1	12
Impatiens glandulifera	5	8	7	7	t	4	1	8
Matricaria discoidea	8	5	7	8	t	1	10	8
Buddleja davidii	8	4	7	4	p	–	–	8
Juglans regia	6	6	7	7	p	–	–	8
Populus canescens	–	–	–	–	p	–	–	8
Populus x canadensis	–	–	–	–	p	–	–	8
Elodea canadensis	7	12	7	7	hy	4	2	8
Heracleum mantegazzianum	9	6	–	8	h	–	–	8
Duchesnea indica	–	–	–	–	–	–	–	8
Claytonia perfoliata	6	5	7	7	t	8	1	4
Taxus baccata	–	–	–	–	p	–	–	4
Aster novi-belgii	9	6	7	9	h	–	–	4
Oenothera biennis	9	4	–	4	h	1	2	4
Oxalis europaea	6	5	5	7	h	1	6	4
Saponaria officinalis	7	5	7	5	h	1	2	4
Fallopia sachalinensis	7	8	7	8	g	1	1	4
Fagopyrum esculentum	–	–	–	–	–	–	–	4

L, F, R, and N are indicator values according to Ellenberg et al. (1991) for important ecological factors, respectively luminosity, humidity, reaction and nitrogen. LF: life forms (t: therophyte; h: hemicryptophyte; hy: hydrophyte; g: geophyte; p: phanerophyte); Ecol. gr. and Ra: ecological groups and rarity level according to Stieperaere and Fransen (1982) (1: very rare; 10: very common); Freq: species frequency (proportion of the grid-cells (n = 25) where the species has been noted). –: no data.

commonness and invasibility could be related to some of the abovementioned ecological factors. Significance is set at $P = 0.05$ when Bonferroni corrected (i.e. the original probability multiplied by the number of comparisons in the sample, to account for increasing probability of finding a significant result by chance alone as the number of comparisons increases).

To discriminate the species that are significant indicators of a site's species richness, commonness and invasibility from those that are not, the Student t-test was performed, comparing the value of these criteria for each cell when the one by one considered species are present or not. Because some grid-cells (edge cells) differ in amount of forest, the quality criteria, environmental parameters and ecological

Table 2. Indicator species identified in the Sonian forest (Brussels).

Species	Proportion of non-native sp.			Commonness index			Species richness/km^2		
	Present	Absent	*P*-value	Present	Absent	*P*-value	Present	Absent	*P*-value
Adoxa moschatellina	6.05	5.50	0.3362	5.58	5.87	0.0067 **	138	100	0.0177 *
Ajuga reptans	5.23	6.09	0.2444	5.68	5.87	0.0491 *	131	94	0.0171 *
Alliaria petiolata	6.39	5.02	0.1314	5.76	5.79	0.3726	139	86	0.0012 **
Anthriscus sylvestris	7.69	5.04	0.0139 *	5.86	5.75	0.2430	117	110	0.3933
Arum maculatum	6.81	4.79	0.0548	5.61	5.91	0.0030 **	129	98	0.0459 *
Calamagrostis epigejos	6.22	5.25	0.2258	5.90	5.68	0.0288 *	124	102	0.1285
Equisetum telmateia	6.57	5.40	0.2517	5.53	5.85	0.0028 **	156	98	0.0072 **
Festuca gigantea	6.27	4.79	0.1202	5.71	5.88	0.0756	130	84	0.0039 **
Galeopsis tetrahit	7.51	4.24	0.0035 **	5.86	5.71	0.0965	133	95	0.0189 *
Hyacinthoides non-scripta	5.79	5.65	0.4679	5.55	5.83	0.0094 **	118	110	0.3170
Impatiens noli-tangere	6.11	5.47	0.3001	5.58	5.86	0.0105 *	146	96	0.0094 **
Luzula pilosa	5.65	5.70	0.4854	5.66	5.86	0.0379	126	101	0.0800
Polygonatum multiflorum	6.25	5.15	0.1871	5.62	5.92	0.0029 **	138	87	0.0015 **
Primula elatior	5.63	5.69	0.4880	5.51	5.84	0.0222 *	144	104	0.0733
Ranunculus ficaria	5.46	6.00	0.3449	5.67	5.94	0.0052 **	127	89	0.0196 *
Senecio ovatus	7.23	4.24	0.0058 **	5.75	5.80	0.3349	138	88	0.0020 **
Viola reichenbachiana	6.32	4.98	0.1351	5.68	5.88	0.0400 *	125	97	0.0591

Present: value of the quality criterion when the considered (indicator) species is present in the sample plot. Absent: value of the quality criterion when the considered (indicator) species is missing in the sample plot. Student *t*-test: $*P \leq 0.05$; $**P \leq 0.01$.

Table 3. Pearson correlation coefficient (r^2) and level of significance between different ecological parameters in 1 km^2 samples of the Sonian Forest in Brussels.

r^2	Exo	Com	N	R	F	L	hy	hel	t	g	h	c	p	Ecol. gr. 1	Ecol. gr. 2	Ecol. gr. 4	Ecol. gr. 5	Ecol. gr. 6	Ecol. gr. 7	Ecol. gr. 8	Ecol. gr. 9	Richness	Asph. roads	Total roads
Exo																								
Com	0.0000																							
N	0.0018	0.1387																						
R	0.0301	0.0483	0.0635																					
F	0.0079	0.0086	0.0994	0.1928																				
L	0.1699	0.0477	0.0348	0.1312	0.1095																			
hy	0.0800	0.0192	0.0040	0.1330	0.2945	0.3140																		
hel	0.0063	0.0495	0.0432	0.1658	0.4260**	0.3131	0.2584																	
t	0.1100	0.0008	0.1302	0.0040	0.0874	0.0941	0.0706	0.0000																
g	0.0575	0.3810*	0.0106	0.0690	0.0014	0.1628	0.0016	0.0051	0.0076															
h	0.2576	0.0868	0.0132	0.0456	0.0089	0.0000	0.1044	0.0038	0.2204	0.2382														
c	0.0114	0.0195	0.0584	0.0885	0.2212	0.1548	0.1772	0.1908	0.0037	0.0123	0.0173													
p	0.2655	0.0081	0.0353	0.0096	0.0160	0.0254	0.0035	0.0495	0.0376	0.0016	0.3186	0.0060												
Ecol.gr. 1	0.1654	0.1704	0.0657	0.0494	0.0154	0.2358	0.0222	0.0312	0.3357*	0.0494	0.0156	0.0383	0.0151											
Ecol.gr. 2	0.1223	0.0117	0.0589	0.0000	0.2568	0.0066	0.0043	0.0003	0.0704	0.0574	0.0908	0.0212	0.0006	0.0385										
Ecol.gr. 4	0.0876	0.0218	0.0104	0.3643*	0.4270**	0.5012**	0.5497***	0.6492***	0.0060	0.0008	0.0206	0.2709	0.0300	0.0015	0.0057									
Ecol.gr. 5	0.0001	0.0018	0.1708	0.1594	0.0499	0.4858**	0.0890	0.2803	0.0591	0.0177	0.0440	0.2929	0.2055	0.0223	0.0016	0.2860								
Ecol.gr. 6	0.0340	0.0002	0.2890	0.0326	0.1639	0.0195	0.0016	0.0190	0.0461	0.0086	0.0015	0.0238	0.0018	0.1180	0.0601	0.0018	0.0123							
Ecol.gr. 7	0.0200	0.2055	0.5922***	0.0475	0.1734	0.0027	0.0014	0.0007	0.1348	0.0477	0.0447	0.0175	0.0049	0.0116	0.0548	0.0063	0.0780	0.1375						
Ecol.gr. 8	0.0083	0.0253	0.1465	0.0675	0.1472	0.1746	0.1901	0.1727	0.0915	0.0387	0.0369	0.1527	0.0570	0.0108	0.0205	0.2761	0.1949	0.1721	0.1147					
Ecol.gr. 9	0.0931	0.0079	0.0549	0.2361	0.0798	0.8051***	0.2999	0.2406	0.1331	0.1305	0.0094	0.1524	0.1116	0.2205	0.0005	0.4309**	0.6797***	0.0351	0.0210	0.1722				
Richness	0.2250	0.0821	0.2127	0.2024	0.0051	0.4546**	0.2953	0.3124	0.1986	0.0013	0.1315	0.0705	0.0057	0.0469	0.0956	0.4250**	0.3444*	0.0475	0.1766	0.1774	0.4863**			
Asph. roads	0.0019	0.0677	0.0183	0.0332	0.2503	0.0001	0.1324	0.0843	0.1089	0.0016	0.0076	0.0175	0.0236	0.0866	0.1527	0.0806	0.0186	0.0011	0.1316	0.0000	0.0009	0.0078		
Total roads	0.0229	0.0026	0.0389	0.0027	0.1288	0.0018	0.0642	0.0058	0.0563	0.0008	0.0155	0.0193	0.0626	0.0299	0.2732	0.0044	0.0249	0.0017	0.0663	0.0001	0.0002	0.0233	0.6823***	

Bonferroni adjusted probabilities (i.e. the original probability multiplied by the number of comparisons in the sample (n = 23) to account for increased chance of finding a significant result as the number of comparisons increases) shown as: * $P < 0.05$; **$P < 0.01$; *** $P < 0.001$. Exo: exotic species; Com: commonness index; N: nitrogen index; R: reaction index; F: humidity index; L: luminosity index; hy: hydrophytes; hel: helophytes; t: therophytes; g: geophytes; h: hemicryptophytes; c: chamephytes; p: phanerophytes; Ecol. gr.: ecological group; Asph.: asphalted.

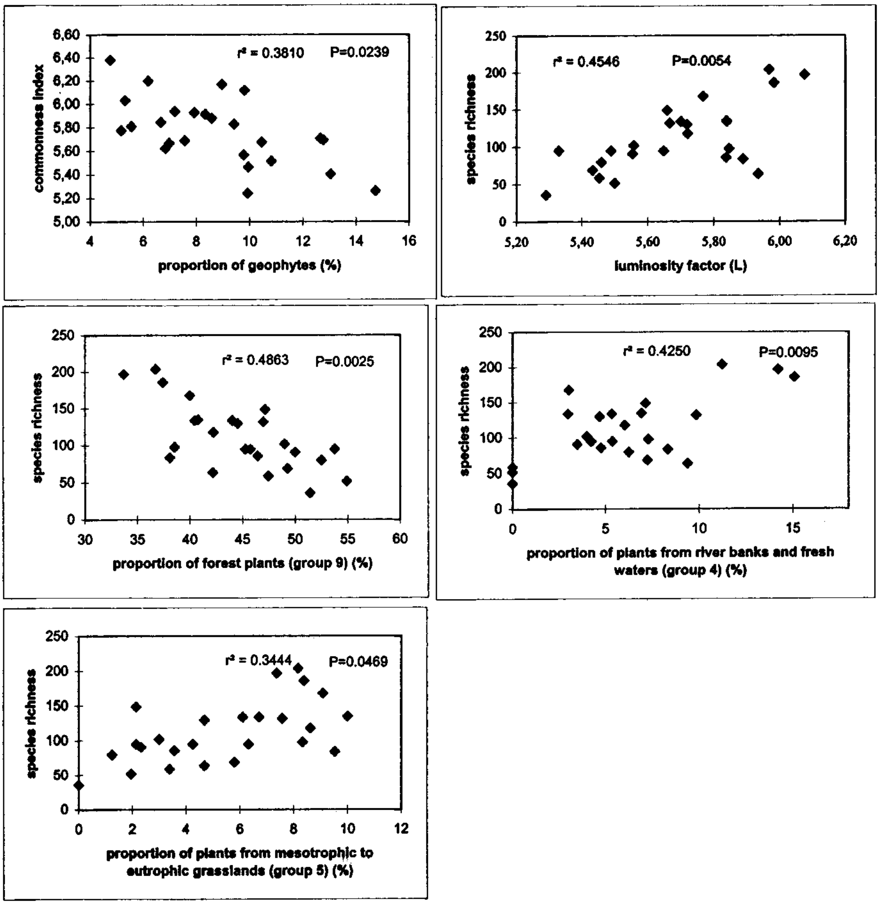

Figure 2. Relationship between criteria of habitat quality (species richness and commonness index) and environmental parameters or species ecological groups, calculated for the Sonian Forest in Brussels. Bonferroni adjusted probabilities are used (i.e. the original probability multiplied by number of comparisons in the sample ($n = 23$) to account for increased chance of finding a significant result as the number of comparisons increases).

groups were standardised, i.e. expressed as relative quantities and not absolute numbers, which would affect the results.

Results

Three hundred and seventy two plant species have been identified in the study area. Most common trees are *Fagus sylvatica, Acer pseudoplatanus, Betula pendula, Carpinus betulus* and *Quercus robur*. Most frequent shrubs are *Sambucus nigra* and *Salix caprea*. Most observed herbaceous species are *Athyrium filix-femina, De-*

schampsia cespitosa, Dryopteris carthusiana, Epilobium angustifolium, Geum urbanum, Luzula sylvatica, Oxalis acetosella, Plantago major, Ranunculus repens, Rumex sanguineus and *Urtica dioica.*

Thirty-three non-native species were found (Table 1). The most common of them is *Impatiens parviflora*, which was censused in 92% of the grid-cells. Other non-native plants which are recognised as invaders are also present, such as *Fallopia japonica, F. sachalinensis, Solidago gigantea, Impatiens glandulifera* and *Heracleum mantegazzianum.*

Among the encountered taxa, 17 have been recognised as indicators of either commonness, species richness or invasibility (Table 2). *Anthriscus sylvestris, Galeopsis tetrahit* and *Senecio ovatus* indicate habitats with a higher proportion of non-native species, i.e. biotopes which are sensitive to invasion. On the other hand, *Hyacinthoides non-scripta, Primula elatior* and *Viola reichenbachiana* are good witnesses to rarer biotopes. At last, *Adoxa moschatellina, Ajuga reptans, Arum maculatum, Equisetum telmateia, Impatiens noli-tangere, Polygonatum multiflorum* and *Ranunculus ficaria* are indicators of richer and rarer habitats.

Table 3 gives the correlations between all the considered parameters (Bonferroni corrected probabilities). Figure 2 points out the significant relationships involving species richness, commonness and invasibility. In the studied area, commonness decreases with increasing number of geophytes. Species richness increases with increasing luminosity factor, proportion of river bank and fresh-water plants, and proportion of plants from mesotrophic to eutrophic grasslands. Inversely, species richness decreases when the proportion of forest plants increases.

Discussion

The indicator idea

In the vast literature concerning indicator species, one finds many different meanings of the term indicator species, including (1) a species that indicates particular environmental conditions (Ellenberg et al. 1991; Kremen 1992); (2) a species thought to be sensitive to and therefore to serve as an early warning indicator of environmental changes such as global warming (Parsons 1991); (3) a keystone species, which is a species whose addition to or loss from an ecosystem leads to major changes in abundance or occurrence of at least one other species (Mills et al. 1993); (4) a species whose presence indicates human-created abiotic conditions such as air or water pollution (Spellerberg 1994); (5) a species that indicates a guild, i.e. a group of species that exploit the same class of environmental resources in a similar way (Root 1967); (6) a management indicator species, which is a species that can be used to monitor the success or failure of management practices to sustain biodiversity (Lindenmayer et al. 2000); (7) a structure-based indicator, i.e. a species that indicates stand- and landscape-level (spatial) features, such as stand structural complexity, connectivity and heterogeneity (Lindenmayer et al. 2000). Noss (1990) recognises several other indicator types in addition to the above, including 'umbrellas' (species with large-area requirements that, if protected, will also protect a variety of species with smaller area requirements), 'flagships' (popular, charismatic species that can attract the attention of the public), and 'vulnerables' (species that are viewed as being at risk).

This paper reports a study of the assessment of site quality with the use of indicator plants. The convenience of using plants as indicators of site quality has been recognised for many years (e.g. Cajander 1926; Braun-Blanquet 1932; Noirfalise 1984; Ellenberg 1988). Mark and Lawesson (2000), for example, identified indicators of high conservation value, as well as indicators of low conservation value of beech forests. Other authors developed the concept of ancient-forest species (Peterken 1974; Wulf 1997; Honnay et al. 1998), that indicate the ecological value of a forest, i.e. sites that are associated with low management impact and long forest continuity.

Fourteen indicators of the quality of the biotope (in terms of species richness and/or commonness) have been identified in the present study. This mainly concerns plants indicating mesotrophic mull, on deep soil with good water reserves. Thus, for the study area, soil reaction and humidity are acting favourably on the species richness and/or the commonness of the habitat. These indicators can be used to evaluate the nature quality across time and space in peri-urban forests in North Belgium. With these indicators, which are rather common species in the study area, any forest compartment could be evaluated, even if none of the rarer species are present. Indeed, as already highlighted by other authors (e.g. Mark and Lawesson 2000), subjective field observations are in many cases the basis for selection of an indicator species, and the species selected in this way are often rare and statistically inappropriate. Therefore, indicators among common species are necessary to achieve a useful evaluation method. Another advantage of our study lies in the use of 'low quality'-indicators, such as invasibility indicators. With the registration of these species, it should be possible to monitor unwanted disturbance effects more rapidly than if changes in the species richness indicators and commonness indicators need to be awaited.

Non-native species

In the present study, we found a proportion of non-native species fluctuating between 1.45 and 13.19%. In North America, Stapanian et al. (1998) found similar results with 4.5 to 13.2%. They underlined the fact that the plot proportion with at least one non-native species is significantly higher in disturbed areas than in non-disturbed ones. Our study shows the same phenomenon, as the proportion of non-native species is higher in edge cells located along the city–forest ecotone (results not shown).

The present study indicates that *Impatiens parviflora*, originating from Asia, is the most frequent non-native in the Sonian Forest, which essentially belongs to the *Fagion* phytosociological group (beech forests). This agrees with Cwiklinski's (1978) results, showing that this non-native species is more abundant in beech forests where it occupies 38% of the surface of the phytocoenosis. Coombe (1956) also related the high gregariousness of the species, forming extensive populations in woodlands. In forests of Central Europe, *Impatiens parviflora* is also the most widespread non-native species (Trepl 1984).

Three native taxa (*Anthriscus sylvestris, Galeopsis tetrahit* and *Senecio ovatus*) were identified as good indicators of sites which are potentially targets for non-native species. These three taxa characterise open habitats (e.g. forest edges or clear-felled areas). This suggests that felling is somewhat responsible for the spread of non-native species. Coombe (1956) also pointed out the fact that *Impatiens parviflora* sometimes dominates gaps caused by felling or wind-throw in woodlands. As Bormann and Likens (1979) pointed out, the bulk of species that will participate in the changes that follow clear-cutting are present, in active or dormant form, in the unaffected forest. The occurrence of buried seeds and their germination after disturbance has been known for a long time (Marks 1974). This plant strategy involves (Bormann and Likens, *op. cit.*): (1) large quantities of dormant seeds in the soil; (2) germination triggered by environmental signals associated with disturbance or light quality; (3) rapid height growth; (4) relatively rapid sexual reproduction with re-establishment of the dormant seed pool. According to these authors, clear-cutting results in both a markedly elevated concentration of NO_3-N in the soil solution and a marked diminution of activity by rootlets of the trees that previously occupied the site (possibly elimination by allelopathic root exudates inhibiting seed germination). So, the question arises as to the responsiveness of species to this temporary condition. Indirect evidence suggests that some species are capable of responding to this condition with greatly accelerated growth rate. But why would non-native species develop better than native plants once the clear-cut is achieved? Many non-native species reach reproductive maturity relatively early, as well as having high seed production. A short juvenile period and a short interval between large seed crops mean early and consistent reproduction and hence rapid population growth (Rejmanek and Richardson 1996). So, probably the early burst of height growth allows the species to gain local dominance quickly, partially shade its competitors and get a proportionately higher share of local site resources. Thereafter, rapid height growth is of less strategic value, since the species has temporarily gained a better competitive position and other species must live under the conditions it creates.

Prediction of invasions and management of vulnerable forest sites

Patterns observed in this study have shown that habitat quality (through criteria like commonness, species richness and invasibility) can be evaluated on the basis of the presence of some indicator species. This can be useful for forest managers and for all persons who are involved in forest planning. Indeed, without having to examine the whole flora in a given place, the areas of most interest (the richest and/or the rarest) and the sites which are the most sensitive to invasion of non-native species can be rapidly identified on the basis of the knowledge of a small number of good indicators. This approach can therefore serve needs in urban forest science, urban forestry planning and management within cities.

Our results allow predictions to be made given specified initial conditions.

Vegetation types containing either *Anthriscus sylvestris, Galeopsis tetrahit* or *Senecio ovatus* were found to be more susceptible to invasion than other habitats. They give a sign that the ecological requirements of non-native species are provided by the habitat. These three species are characteristics of disturbed areas. This observation is consistent with other studies (e.g. Hobbs and Huenneke 1992; Stapanian et al. 1998) indicating that the presence of non-native species is correlated with anthropogenic disturbance, which affects the structure and composition of both the ground layer and upper canopy of forest habitats. Therefore, management strategies that aim to suppress human impact should help to prevent the installation and invasion of non-native species.

Some authors recognise that invasions have already been responsible for a large proportion of all the extinction that is known to have occurred during the historical period (e.g. Macdonald et al. 1989). Thus, locating stands having a high risk of invasion with the help of native flora offers an interesting perspective for a preventive control policy of non-native taxa in place of a curative control policy, as we tried to do until now. An overview of the distribution of *Anthriscus sylvestris, Galeopsis tetrahit* and *Senecio ovatus* in the studied territory would allow us to draw up the potential area of non-native species in the forest. So, we might be able to have a precise view of risk areas needing inspection and management in order to minimise odds of establishment of a non-native flora. This type of preventive control would be more efficient than a curative policy, which often gives poor results. Indeed, the few significant attempts that have been made to control invasive organisms for the purpose of maintaining biodiversity have generally been too little or too late (Macdonald 1994). Nevertheless, the importance of detecting non-native invasions early and initiating appropriate levels of control immediately, has repeatedly been stressed (Usher 1989; Dickson 1998). It is often observed that spread may be delayed and that an introduced plant initially occurs at low levels in its new environment and later suddenly undergoes a population explosion (Cronk and Fuller 1995), and it is now widely recognised that a successful invasion often facilitates further invasions (Cheke 1987). This is important in view of the recent invasion of some non-native species in Brussels, as pointed out by Godefroid (1996b, 1998), and because plant extinctions could be accelerated by invasions in the coming years if control strategies are not effectively implemented (Macdonald 1994). It is not only important for biodiversity conservation but also for forest management. Indeed, many investigators have suggested that introduced herbaceous plants are successful competitors against native woody species (D'Antonio et al. 1998), as competition for water (Eissenstat and Caldwel 1988) or light (Hughes and Vitousek 1993) is important.

In this paper, prediction of non-native invasions is facilitated by the association of particular species with some native plants. This prediction technique might be used to identify and prioritise sites for protection against future invasions. Our indicators are able to detect a condition far enough in advance to assist in solving the problem, which is a condition expressed by Dallmeier and Comiskey (1998) for a good indicator. A major issue is how much human resources should be expended in an effort to eliminate present invasions and prevent future invasion. Biological, mechanical or chemical control are possible but, as peri-urban forests are strongly influenced by the nearby city, an efficient control policy should be applied over the whole area of the city, which is the most important source of propagules. This would require an expensive long-term commitment that, to date, has not been given serious consideration by the managers who would have to pay the bill. However, the costs of allowing invasions to proceed are uncertain and difficult to evaluate. We suggest that a cost-effective way to prevent the transformation of forest patches into alien-dominated forests is to survey the vulnerable sites frequently and eliminate seedlings of non-native species before they spread. Prospects for eradicating all non-native plants from the Sonian Forest are not encouraging, but at least there remain forest parts that are free of non-indigenous species. With appropriate survey and minimal expense, it will be possible to protect them from future invasion.

Conclusions

This paper aims at promoting the use of indicators in environmental management. Results indicate a great potential for increasing herb-related benefits through broadened urban forestry management. Forest planning could and should take into consideration the herb layer. This would allow areas utilised for commercial forestry to serve more effectively a multiplicity of uses and needs, including those of interest for forest managers, conservationists and recreational planners. An interesting perspective for a preventive control policy of non-native taxa is the location of stands having a high risk of invasion with the help of native species. This is particularly interesting because an effective control of well-established non-native organisms is becoming extremely difficult. Our results could be used as a basis for deducing the responses of the ecosystem to the environment and to generate certain principles and hypotheses that can be applied in informed urban management in other cities of Western Europe.

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