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Carabid beetles in green infrastructures: the importance of management practices for improving the biodiversity in a metropolitan city

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Abstract Green infrastructure in urbanized areas has a dual purpose that achieves both sociological and ecological goals. To benefit society and the urban ecosystem, green infrastructure should be effectively managed. We investigated carabid diversity and assemblage structure as indicators of biodiversity in green infrastructures in a rapidly developing urban area to identify the habitat's values. In addition, we attempted to reveal the effect of environmental variables (e.g., vegetation structure, soil, and disturbance) that strongly contribute to carabid diversity and assemblage structure. Of the collected organisms, 6,154 individuals representing 20 carabid species were identified in the green infrastructure. Those species with flight ability and that were found in dry habitats were widely distributed and dominated green infrastructures. Carabid assemblages changed significantly in response to management practices. These changes were both positive and negative, with the negative changes increasing the instability of the carabid assemblages through the destruction of their habitat. Other factors, such as the presence of original habitat, habitat age, and habitat succession, also had a considerable effect on carabid diversity. We revealed that management practices prevented habitat succession, and these interactive effects determined carabid diversity and structure in green infrastructures.

Keywords Abandonment area · Carabid beetle · Green infrastructure · Management practice · Spontaneous succession · Urbanization

Introduction

Urbanization is a dominant demographic trend and an important component of human societies and in the alteration of natural ecosystems. The human population will continue to aggregate in urban areas and the structure and function of natural ecosystems will be continuously influenced by urbanization (Foster et al. 2003; Pickett et al. 2008). Although

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urban areas are primary human habitats, urban society has been recognized as playing an important role in the function of the remaining semi-natural ecosystems in urban areas, particularly when providing ecosystem services, improving human well-being, and fulfilling ethical responsibilities (Gill et al. 2007; Daily et al. 2009). This is the greatest motivation for conserving biological diversity in urban landscapes (Dearborn and Kark 2009). Degraded biological diversity must be restored through the conservation of nature at the fringes of habitats and the restoration of remnant patches of natural ecosystems. Working toward these goals may provide the opportunity to recover and redevelop the connection between humanity and nature in urban landscapes (Savard et al. 2000; Tzoulas et al. 2007).

Studies have suggested management practices for the conservation of habitats that target urban habitat size, location, and the degree of connectivity of each habitat, all of which are important environmental factors influencing biodiversity in urban habitats (reviewed by Gill and Bonnett 1973; Gilbert 1989; McIntyre 2000; McKinney 2008). In fact, an approach using the aforementioned information and evidence could be very efficient and effective, even though most management practices are based on experience gained in natural habitats and agricultural areas. However, in a city that is young, with a rapidly evolving urban footprint that has expanded dramatically over the past few decades, urban habitats require management by more unconventional methods (Ramalho and Hobbs 2012). Moreover, some young urban habitats are needed to identify their ecological roles and the characteristics for improving the biodiversity in urbanized areas (Colding et al. 2006; Seastedt et al. 2008; Goddard et al. 2010).

In this study, we attempted to identify the biodiversity in urban habitats, particularly green infrastructures, in a young and rapidly developing urban area. Green infrastructures such as parks, roadsides, and gardens, as well as derelict sites and wastelands, are potential habitats for wildlife. Increasing green space (as new, temporal, and artificial habitats) and its associated biodiversity, is a means to improve existing green space (Angold et al. 2006). In addition, relationships between biodiversity and environmental factors of the urban habitats were investigated in order to identify the environmental factors characterizing and affecting the biodiversity of urban habitats (Blake et al. 1996; Niemelä and Kotze 2009).

Within urban centers, carabid beetles serve an important role as indicator species in assessing the impacts of urbanization (reviewed by Niemelä and Kotze 2009). Carabid diversity composition and traits change along an urbanization gradient, indicating the different effects of increasing urbanization—members of this family are very sensitive to natural and anthropogenic disturbances at both the habitat and the landscape levels (Koivula 2011). This study thus addresses three questions: (1) Are green infrastructures characterized by a typical suite of carabid beetles in urbanized areas? (2) Are management practices for green infrastructures an important driving factor for carabid assemblage composition? (3) Which environmental variables characterizing green infrastructure (e.g., vegetation structure, soil, and previous habitat form) contribute to the maintenance of carabid beetles in this habitat?

Material and methods

Sampling sites

This study was conducted in southwestern Busan Metropolitan City (area=767.35 km^2, population density of the city=4,692/km^2, urbanized area 29.25 %, road pavement=96.84 %). Within the city remnant green spaces were assessed based on their role within the urban system. Eight types of urban infrastructures were identified; park, closed landfill, restored urban wetland, non-managed grassland, brownfield, garden inside interchange, roadside, and forest park in an urbanized area. Study sites were chosen according to their ability to represent carabid beetle diversity and the effect of management practices on carabid assemblages (Table 1). To further assess the impact of urban development and management practices it was necessary that these urban infrastructures be clearly isolated from natural habitat areas.

The park was created in 2006 in an area that was formerly agricultural land. The parking lot and public facilities were converted and planted with *Pinus densiflora* (Siebold and Zucc.) and grass. For visitor convenience, the plants are mowed three times a year. The landfill, previously agricultural land during the 1980s, was converted to a landfill in 1993 and filled with household waste. In 1997, the landfill was covered with roughly 0.9 m of sand and sealed off with clay till. After closing, the landfill was no longer managed and was left to secondary succession. The urban wetland was restored in 2006 after 40 years of agricultural use. *Phragmites communis* Trin. and *Salix* sp. were planted and came to dominate this habitat. Citizen access is often restricted to conserve the wetland habitat. The non-managed grassland is located in a riparian corridor. This was also formerly barren land until the 1990s, when it was acquired to secure green space in an urbanized area, although wide areas of the corridor were developed as a riverside park. A brownfield is located in an apartment area that was constructed on reclaimed land in 2006. The brownfield area had been prepared for residential construction but the construction plans were abandoned in 2008. A garden was created within an interchange surrounded by an eight-lane road. *P. densiflora* (Siebold and Zucc.) and grasses were planted here as they were with the park. The roadside along the right-of-way is approximately 2 m wide and 1 km long and was planted with trees to separate the road from the residential area. To improve visibility for motorists, the grass areas inside the interchange and along the roadside are mowed twice a year. The forest park is fragmented by urban development and surrounded by residential areas. Although this forest park is a privately owned area, it is open to the public. A *P. densiflora* (Siebold and Zucc.)—*Quercus* sp. community, which was planted in 1953, dominates the forest park. The forest park is regularly cleared of trash and mowed; however, these activities are not intensive.

Carabid sampling

Carabid beetles were sampled using pitfall traps (38 cm in diameter, 15.5 cm deep) that were partially filled with a propylene glycol–water mixture (50:50). Three traps were installed at

Table 1 Characteristics of urban habitats for this study

Types	Origins	Disturbance (Type)	Habitat age[a]	Types[b]
Park	Agricultural land	Mid (Mowing, trampling)	6	Nature 3
Closed landfill	Agricultural land Landfill	Low (None)	15	Nature 4
Restored urban wetland	Agricultural land	Low (None)	6	Nature 3
Unmanaged grassland	Bared land	Low (None)	6	Nature 4
Brownfield	Derelict area	Low (None)	4	Nature 4
Garden inside interchange	Agricultural land	High (Mowing)	15	Nature 3
Roadside	Urban	High (Mowing, trampling)	15	Nature 3
Forest park	Forest	Mid (Trampling, clean up)	50	Nature 1

[a] Habitat age, Years since change in type of infrastructure; [b] Nature 1, Old wilderness; Nature 2, Traditional cultural land; Nature 3, Functional greening; Nature 4, Urban wilderness (modified from Werner and Zahner, 2010)

each site along a linear transect at 100-m intervals. As far as possible, the traps were installed along a central axis in homogenous vegetation stands at each site. The trapping period covered most of the growing season (May 3 to November 23, 2012), and traps were emptied once a month. The ecological traits of carabid beetles are closely correlated with environmental factors. Specially breeding seasons are reflected the environmental condition. Some carabid beetles showed the different breeding and appearance patterns in seriously disturbed habitat. Additionally, the flight ability and food needs are important factors affecting population density and diversity in isolated and disturbed habitats (Raino and Niemelä 2003; Fattorini 2011). The ecological characteristics (e.g., breeding season, habitat preference, feeding type, and flight ability) of each collected carabid species for all study sites were obtained from Do et al. (2011, 2012b) and the Working Group for Biological Indicator Ground Beetles Database (2011). Each species was categorized according to preferred habitat (grass/forest), breeding season (spring/autumn), and flight ability (capable of flight/flightless; a designation based on the presence of flight wings).

Environmental variables

We assessed the environmental variables associated with carabid beetles and habitat condition, including year of habitat construction/development (i.e., habitat age), disturbance levels, previous habitat form (i.e., habitat origins), vegetation structure along with plant species richness, coverage by different layers, and soil texture.

Habitat ages were identified by the year the habitat was created and/or constructed. Habitat origins were identified using aerial photographs taken before creation and/or construction and other documents. Total disturbance levels were considered based on the frequency of management practices (e.g., mowing), number of visitors, and accessibility. The vegetation structure and soil texture were identified for assessment of habitat characteristics. Vegetation structure was described by estimating the proportional cover of tree-layer, sub-tree-layer, and grass-layer plant species for each site. Vertical stand structure was assessed using a visual cover method recognizing three vegetation strata: herb layer (10 cm–1 m), shrub layer (>1–2 m), and tree layer (>2–10 m). Shrubs and trees were sampled using 10 by 10 m plots and herbaceous vegetation was sampled using 1 by 1 m plots. The number of plant species was identified for each plot. Soil samples were taken at 5 to 10 cm depths for each site. These samples were used to establish the soil texture using an LS 13 320 laser diffraction particle size analyzer (Beckman Coulter Inc., Brea, CA, USA). The volume percentage of clay, silt, and sand were calculated with Beckman Coulter LS Software.

Data analysis

Carabid species richness for the different urban habitat types was compared using rarefaction curves. All samples of a particular habitat were pooled together for this analysis. Individual-based rarefaction analysis gives an estimation of the expected number of species while taking into account sampling effort for the total number of catches. By showing the rate of new species accumulation, rarefaction curves indicate whether sufficient samples were collected to make valid comparisons. Rarefaction curves were generated using PAST (Paleontological Statistics; Hammer et al. 2001). Non-metric multidimensional scaling (NMDS) was used to ordinate differences in community composition using Ward's clustering method calculated using Bray–Curtis similarity distances for urban habitats. This analysis was undertaken in PC-ORD version 6 (McCune et al. 2002). Thus, Simpson's dominance index (D′; Simpson

1949), Shannon's diversity index (H′), and Shannon's evenness index (J′; Shannon and Weaver 1963) are calculated to assess species diversity using PAST.

A generalized linear model (GLM) was used to analyze the data of carabid richness and abundance per site and for each ecological trait (e.g., breeding, type of habitat occupied, flight capacity) per habitat group or habitat condition. The hypothesis that habitat condition has an effect on carabid richness, abundance, and ecological traits was tested across the different habitat groups. A post hoc Tukey test was performed for multiple comparisons, where statistical significance was determined at $\alpha=0.05$. These analyses were performed using PASW Statistics 18.

Patterns of species richness and abundance were visualized by plotting the log abundance of ranked species in decreasing order of abundance (MacArthur 1960; Whittaker 1965). These plots of ranked species abundances can reveal species assemblage structures specifically to identify the effects of disturbance on the assemblage (Magurran 1988). Therefore, to identify the effects of the environmental variables of each habitat on carabid beetles, rank-abundance plots of carabid species were created for different carabid assemblages that were identified using NMDS clustering methods. These plots were fitted individually to the different models, including null model (Broken Stick), preemption (geometric), lognormal, Zipf, and Zipf-Mandelbrot distribution (Magurran 2004; Ugland and Gray 1982). "Best Model" selection was based on the difference of Akaike's information criterion (AIC) and Bayesian information criterion (BIC) values for the compared models (Kindt and Coe 2005). These analyses were performed using BiodiversityR (Kindt and Coe 2005), which was developed for the R 2.1.1 statistical language and environment (R Development Core Team 2008).

Most environmental variables were correlated, which may result in multicollinearity of environmental responses and a bias in parameter selection. To limit the effect of multicollinearity and reduce the number of variables to enable more concise management plan development, the variables can be replaced by their principal component scores (Legendre and Legendre 1998). This procedure is applicable when the variables, which may not have a strong influence independently, but when combined have far more important effects. To reduce the effect of the combined variable interactions, factor analysis (FA) using principal components extraction was performed with a VARIMAX normalized rotation on the environmental variables using PASW Statistics 18. The two principal components were retained and the loadings examined to identify their relationships to the original variables. The principal components were used in a stepwise regression analysis to identify the correlation among environmental variables and total abundance, richness, Shannon's diversity and evenness, and Simpson's dominance.

Results

Carabid diversity

We collected 6,154 adult carabid specimens representing 20 carabid species from urban green infrastructures (Table 2). Approximately 65 % of the carabid species were collected from restored urban wetlands and forest parks. Three carabid species, *Dolichus halensis*, *Anisodactylus signatus*, and *A. punctatipennis*, made up 46.23 % of the total catch. *A. signatus* (31.79±7.17 mean individuals in each habitat±S.E.) and *A. punctatipennis* (31.88±8.70) were widely distributed throughout the urban habitats, with large populations in each site (24 sites). Thirteen species of autumn breeders, which produce summer larvae, and seven species of summer-breeding carabid beetles were recorded. Most species (17 species,

Table 2 Carabid beetles and their diversities in different urban habitats

Species name	Abbr.	EcolT	P	CL	RUW	NG	BF	GI	RS	FP
Nebria livida angulata	Nli	A/G/N	0	4	25	0	0	0	0	0
Dolichus halensis	Dha	A/G/N	216	350	324	432	0	0	0	108
Synuchus nitidus	Sni	A/F/N	0	0	0	0	0	0	0	288
Synuchus agonus	Sgo	A/F/N	0	0	0	0	0	0	0	360
Amara chalcites	Alu	A/G/F	0	81	7	80	109	0	0	132
Amara macronota	Ama	A/G/F	14	54	11	56	54	0	0	192
Anisodactylus signatus	Asi	S/G/F	81	52	40	122	65	48	45	324
Anisodactylus punctatipennis	Apu	S/G/F	108	62	14	102	43	41	21	384
Harpalus capito	Hca	A/G/F	4	108	54	50	10	14	0	132
Harpalus sinicus	His	A/G/F	9	54	40	24	2	0	0	72
Harpalus tschiliensis	Hts	A/G/F	0	14	0	0	0	0	0	108
Lesticus magnus	Lma	S/G/N	38	94	16	0	0	0	0	0
Colpodes japonicas	Cja	A/G/F	0	0	0	0	1	0	0	192
Colpodes buchanani	Cbu	A/G/F	0	0	0	16	0	2	0	264
Colpodes adonis	Cad	A/G/F	0	0	0	0	0	0	0	84
Chlaenius naeviger	Can	A/G/F	0	4	0	62	2	0	0	0
Haplochlaenius costiger	Hco	S/F/F	60	108	54	0	22	0	0	0
Brachinus stenoderus	Bst	S/G/N	0	0	3	24	0	0	0	0
Pheropsophus javanus	Pja	S/G/N	0	0	82	12	0	0	0	0
Pheropsophus jessoensis	Pje	S/G/N	0	0	14	0	0	0	0	0
Abundance			8	12	13	11	9	4	2	13
Richness			530	985	684	980	308	105	66	2640
Shannon's H′			1.62	2.04	1.85	1.85	1.66	1.07	0.63	2.43
Shannon's J′			0.63	0.64	0.49	0.58	0.58	0.73	0.93	0.88
Simpson's D′			0.25	0.18	0.26	0.24	0.23	0.38	0.57	0.10

Abbr. abbreviation, *EcolT* ecological traits, *P* park, *CL* closed landfill, *RUW* restored urban wetland, *UG* non-nmanaged grassland, *BF* brownfield, *GI* grassland inside interchange, *RS* roadside, *FP* forest park

85 %) occurred in grassy habitat rather than in forest habitat. Twelve of the identified species possessed full wings for flight (Table 2).

Rarefaction curves for the forest park, restored urban wetland, non-managed grassland, park, and roadside reached a plateau, indicating that there was sufficient sampling. In contrast, rarefaction curves for the brownfield and garden inside the interchange did not reach an asymptote. Rarefaction curves revealed that non-managed and/or stable habitats had higher than expected carabid richness based on independent sampling efforts (Fig. 1).

Carabid assemblage composition

The two axes in the NMDS ordination accounted for 59.82 % of the variance in species composition (r^2: axis 1=0.723, axis 2=0.167) based on the 20 most commonly occurring species among the 24 study sites (total stress=8.52; Fig. 2). The closed landfill, restored urban

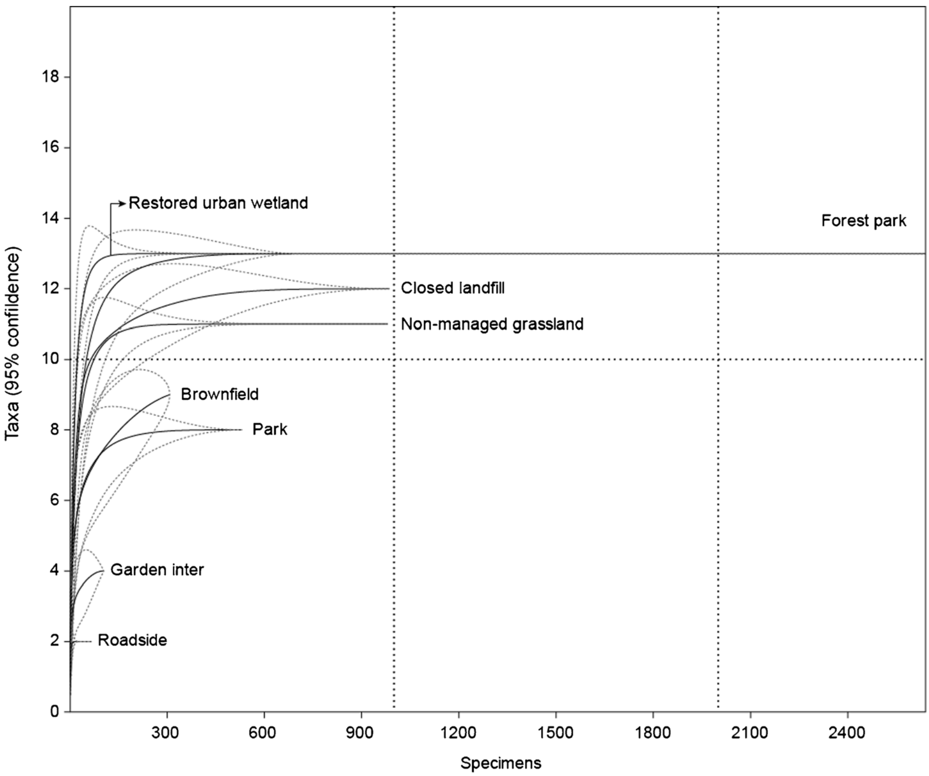

Fig. 1 Rarefaction estimates of expected number of carabid species with standard deviation, by sub-sample size (number of individuals) for total number of carabids collected within each habitat type

wetland, non-managed grassland, and park, which were characterized by a limited disturbance level (Group A), had the lowest scores on axis 2 and exhibited greater separation from the forest park and other intensively managed and/or highly disturbed habitats, including the brownfield, the garden inside the interchange, and the roadside (Group B). In addition, Group B is distinguished from the forest park site on axis 1. Although park sites exhibited comparatively high disturbance, carabid assemblages for the parks were similar to those of the restored urban wetland site because these sites had identical origins as agricultural land, specifically rice-paddy fields.

The rank-abundance plots suggested that the carabid assemblages from Group A were representative of undisturbed habitats, with lognormal distributions (Fig. 3). Carabid assemblages from forest park sites were characteristic of heterogeneous and species-rich environments as described by Zipf-Mandelbrot models. In contrast, carabid assemblages from Group B were fitted by preemption (geometric series) models, indicating unstable conditions characteristic of species-poor and disturbed habitats.

Species richness differed significantly among the habitat groups identified by NMDS (F=32.07, P<0.001). Species richness in Group A was higher than that in Group B (Tukey's HSD, P<0.001), and the difference between species richness in Group A and that of the forest sites was not significant (Tukey's HSD, P=0.49). Carabid abundance differed significantly among habitat groups (F=17.77, P<0.001) although that of Group A was not significantly different from that of Group B (Tukey's HSD, P=0.61). GLM revealed a highly significant

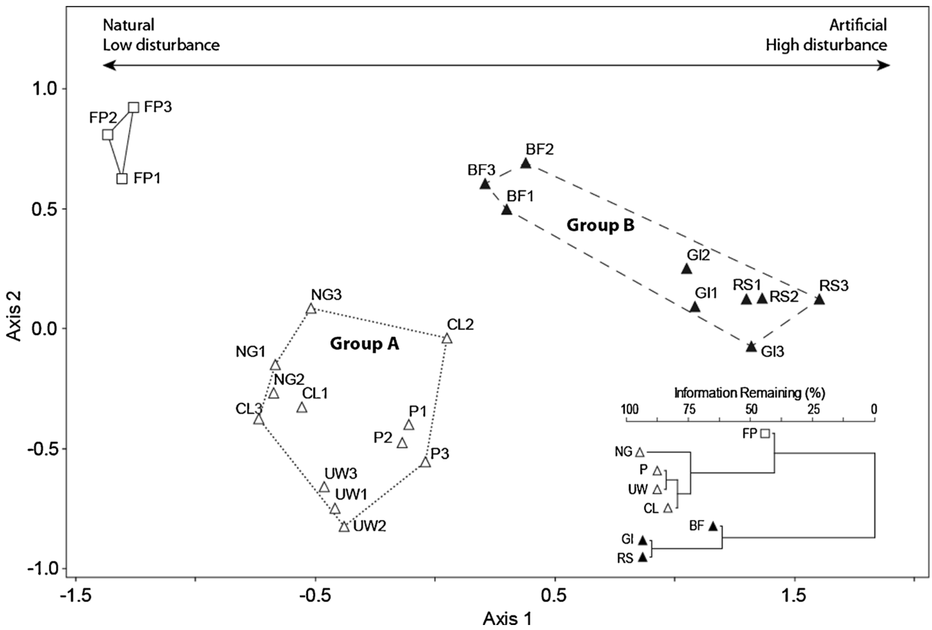

Fig. 2 NMDS ordination based on carabid beetle data from different urban habitats: *P* park, *CL* closed landfill, *UW* restored urban wetland, *NG* non-managed grassland, *BF* brownfield, *GI* grassland inside interchange, *RS* roadside, *FP* forest park

effect of habitat condition (disturbance level) for carabid richness and abundance (Table 3). Carabid richness for each ecological trait within groups differed, whereas richness between groups did not. In contrast, with the exception of breeding types within groups, carabid abundances for species that had different ecological traits differed significantly between and within groups.

Influence of environmental variables on carabid assemblages

On the basis of the results of the PCA, F1 summarized variables according to habitat stability and/or naturalness such as shrub layer, plant species richness, habitat age, habitat origin, and time since development. F2 summarized variables reflecting the degree of disturbance

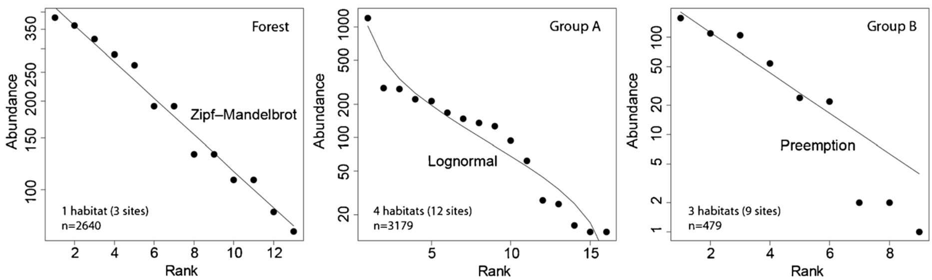

Fig. 3 Rank abundance plots with best fitted distribution models of carabid beetles in the groups of urban habitat resulted from NMDS on a logarithmic scale against species rank in order from the most to the least abundant species

Table 3 GLM table showing the effect of habitat management on (a) carabid richness and (b) abundance of each ecological character

Source variance	SS	df	MS	F	P
(a) Species richness					
Habitat groups	137.15	2	68.57	1.807	0.304
Breeding types within groups	127.06	3	42.35	20.62	<0.001
Habitat groups	137.146	2	186.99	0.37	0.72
Habitat types within groups	629.06	3	209.68	95.44	<0.001
Habitat groups	137.15	2	68.57	0.81	0.524
Flight types within groups	285.67	3	95.22	30.37	<0.001
(b) Individuals					
Habitat groups	60837.81	2	30418.91	25.67	0.009
Breeding types within groups	3758.42	3	1252.81	1.98	0.116
Habitat groups	67767.43	2	33883.71	10.98	0.038
Habitat types within groups	10170.15	3	3390.05	5.48	<0.001
Habitat groups	68232.08	2	34116.04	12.91	0.030
Flight types within groups	8674.01	3	2891.34	4.65	0.003

(Table 4). Multiple regression of carabid diversity against environmental variables for each site gave significant results (Table 5). F1 was positively affected by increasing carabid diversity, particularly abundance, Shannon's H′, and Shannon's J′. In contrast, Simpson's D′ was negatively correlated with F1. The shrub layer, plant species richness, and vegetation coverage strongly influenced increasing carabid diversity. In addition, previous habitat form (e.g., agricultural land and/or natural habitat) had a positive effect on carabid diversity. In contrast, F2 was significantly negatively influenced by carabid richness and Shannon's H′. However,

Table 4 Results of a factorial analysis (FA) using principal components analysis (PCA) on environmental variables. Factor loadings of the first two factors extracted by FA using PCA applied to environmental data

	Factor 1	Factor 2
Eigenvalues	4.29	2.60
Total variance (%)	35.79	21.68
Cumulative eigenvalues	4.29	6.89
Cumulative %	35.79	57.47
Factor loadings		
Grass species richness	0.192	−0.747
Grass coverage	−0.545	−0.663
Sub-tree species richness	0.932	−0.049
Sub-tree coverage	0.944	−0.005
Tree species richness	0.92	0.231
Tree coverage	0.194	0.858
Habitat age	0.78	0.466
Disturbance levels	−0.032	0.744
Origins	0.822	−0.183
Clay	0.088	0.014
Silt	−0.115	0.022
Sand	0.105	−0.074

Table 5 Results of stepwise multiple regressions of carabid diversity on the two factors summarizing the environmental variables extracted by factor analysis

	Richness	Abundance	Shannon's H′	Shannon's J′	Simpson's D′
Regression summary	R^2=0.38	R^2=0.69	R^2=0.57	R^2=0.37	R^2=0.60
	F=6.46	*F*=23.1	*F*=13.75	*F*=6.29	*F*=15.55
	P=0.007	*P*<.001	*P*<0.01	*P*=0.07	*P*<0.01
	SEE=3.32	SEE=155.87	SEE=0.38	SEE=0.15	SEE=0.09
Intercept	B=8.17±0.68	B=256.04±31.82	B=1.59±0.08	B=0.74±0.03	B=0.29±0.02
	P<0001	*P*<0.001	*P*<0.001	*P*<0.001	*P*<0.001
F1	β=0.27±0.69	β=0.80±32.50	β=0.38±0.08	β=0.39±0.03	β=−0.33±0.02
	P=.138	*P*<0.001	*P*=0.015	*P*=0036	*P*=0029
F2	β=−0.56±0.69	β=−0.21±32.50	β=−0.65±0.08	β=0.47±0.03	β=0.70±0.02
	P=0.004	*P*=0.095	*P*<0001	*P*=0.012	*P*<0.001

β standardized regression coefficient (weight)±SE, *SEE* standard error of the estimate

evenness (J′) and dominance (D) indices increased significantly with increasing F2. Disturbance levels and tree cover were the main contributors to F2.

Discussion

In this study, green infrastructures in urban areas supported a variety of carabid beetles, and their assemblage composition changed as the type of green infrastructure changed. For carabid beetles to survive and reproduce, they need a protected or undisturbed space in which to overwinter, mate, and lay eggs. This habitat should also provide food, a favorable microclimate, and shelter from predators (Thiele 1977). Green infrastructures could provide suitable conditions for carabid habitats. Specifically, the forest park supported a different carabid assemblage with high richness and abundance. Although the forest park is artificial and isolated by surrounding urban development, it could act as a significant biodiversity reservoir for an urbanized area. Many previous studies on the effect of urbanization on carabid beetles, specifically those adopting a gradient approach (urban–rural gradient), revealed that species richness and abundance of carabid beetles decreased along an urbanization gradient (Niemelä et al. 2002; Varet et al. 2011). However, in rapidly growing urban areas having a complex, nonlinear, and dispersed structure, using a linear gradient approach may not accurately represent community structure. Koivula and Vermeulen (2005) pointed out that forest beetles' migration between isolated patches was rare due to city highways, as species migration frequency often determines species persistence in fragmented landscapes (i.e., metapopulation structure), this may be an important factor (Fattorini 2013). More directly, Soga et al. (2013) recently revealed that carabid beetle communities in small urban remnants suffered more from "edge effects" than those of large remnants. Thus, small woodlands would be able to maintain only species-poor communities. Furthermore, Do et al. (2012a) suggested that although the forest was fragmented and isolated from surrounding land-use types, carabid richness and community structures showed no difference relative to undisturbed forest areas despite the fact that carabid beetle abundance was lower than in other forested areas. Management practice types and intensities in urban forests could have a stronger, more direct effect on carabid beetles than those practiced in non-urban settings (Taboada et al. 2006; ElSayed and Nakamura 2010). In grassy urban infrastructures (i.e., park, closed landfill, non-managed grassland, and restored urban wetland) management practices (i.e., mowing) strongly influenced to carabid community assemblages. Carabid beetle communities in grassy green infrastructures consisted of a mixture of generalist and open, dry-habitat species with flight ability. *Amara* and *Anisodactylus* were numerous and widely distributed. These findings are similar to the results from brownfield and intensively managed grasslands in urban areas (Eversham et al. 1996; Eyre et al. 2003). In highly disturbed and isolated habitats, the frequency of carabid species with better dispersal abilities was greater than that in undisturbed habitats, and small and generalist carabids such as *Amara* and *Anisodactylus* were widely distributed (Herkert 1994; Ribera et al. 2001). Small et al. (2002) found that small, macropterous species could take advantage of invasion and survive in early successional urban habitats after artificial disturbance, such as soil movement, trampling, and dumping. However, intensively managed habitats, such as the garden inside the interchange and the roadside, had few carabid species. This is similar to results obtained by Hartley et al. (2007) who found that managed grassy habitats in urban settings had less carabid species than surrounding rural unmanaged grassland areas.

We found that original habitats indicative of previous habitat types (e.g., agricultural land and urbanized areas) were significantly related to carabid diversity and community structure in green infrastructures. Carabid beetles in green infrastructures, particularly in sites that were formerly agricultural land, were similar to those of other green infrastructures that had also been converted from agricultural lands. Soil related to habitat origin might be very important in planning and constructing green infrastructures (Pavao-Zuckerman 2008). When doing so, cover soil may be delivered from other areas, particularly from derelict sites. These heterogeneous soils have a significant effect on species directly and indirectly inhabiting green infrastructures (Strauss and Biedermann 2006; Topp et al. 2010). Moreover, the cover soil and its aggregates are related to drainage and soil compaction. Furthermore, soil texture influences the mortality of eggs, larvae, pupae, and imagoes, which, in turn, determine carabid diversity and distribution (Tietze 1987; Brose 2003). Therefore, the soils must also be similar to those found in the surrounding areas or similar species may not successfully establish.

In our study, we found that among environmental variables, the vegetation structure characterizing green infrastructure had relatively little effect on carabid beetle community structure, despite the fact that in many semi-natural habitats, vegetation strongly influences carabid diversity and community structure (Luff et al. 1992; Pinna et al. 2009). When constructing green infrastructures, planting and gardening is of vital importance in achieving environmental, social–behavioral, or aesthetic goals; therefore, the resulting green infrastructures usually appear typologically perfect, but their functional structure (i.e., biodiversity) often fails to meet expectations (Jim 2004; Wu 2008). In contrast, the shrub layer has a more significant effect on carabid diversity. The shrub layer is representative of complex habitat vegetation and is associated with habitat succession. In our study, we found that habitat age was indicative of successional stage, which, in turn, was strongly correlated with carabid richness and abundance in green infrastructures (Mortimer et al. 2002). Although mowing and some other management practices are not intensive, management has deterred initial succession as well as prohibiting the enhancement of biological diversity in green infrastructure (Prach and Walker 2011). Our study showed that these management practices altered the stability of carabid community assemblages. The carabid assemblages of managed green infrastructure were fitted to a geometric distribution. When one abundant species monopolizes a fraction of the resources in proportion to its abundance, the result is a situation generally leading to resource exhaustion (Brown 1984). Instead, carabids from non-managed green infrastructures exhibited mature communities that had reached a stable equilibrium where resources were more evenly consumed among species (Wilson 2009).

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