

Relevance of using a vegetation-based method to conserve urban carabid diversity

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Abstract Urban habitats harbour considerable biological diversity. Ecologists have developed methods to select which habitats should be conserved. The *Ecological value*, a method based on vegetation, has been created for the urban habitats of Montreal (Quebec, Canada). The main objective of our study was to determine if this method was relevant to assess carabid diversity of Mount Royal Park. This index is calculated using five criteria: uniqueness, representativeness, degree of succession, richness and rarity of the flora, each of which can influence communities of insects. Ground beetles (Coleoptera: Carabidae) were selected because of their success as biological indicators. Despite sampling difficulties, our results demonstrate clearly that the *Ecological value* method does not represent the high carabid richness of urban open habitats (tall grasses) and their specialised native carabid species. Within forests we found nonetheless that the ecological value index has a significant positive relationship with native carabid abundance. Moreover, maturity and structure of urban forests were positively correlated with carabid abundance and richness. Some urban vegetation

characteristics have been shown to influence entomological diversity, but the relevance of using a global floral index to encompass the carabid community seems limited.

Keywords Urban tall grasses · Urban forests · Floral index · Ecological value · Native carabids

Introduction

Ecological classification methods can help to improve ecological management of landscape and to achieve better biodiversity conservation planning. In the context of the city of Montreal (Quebec, Canada), a floral index was created in order to select which habitat should be preserved (Domon et al. 1986; Cogliastro et al. 1996). This vegetation-based method, the *Ecological value*, is used to compare and rank the quality of several selected sites. Five criteria are used in the calculation of the ecological value index: *Uniqueness*, *Representativeness*, *Degree of succession*, *Plant specific richness* and *Plant specific rarity* (Table 1). Since heterogeneity of vegetation structure and complexity of landscape mosaic are extreme in urban areas (Rebele 1994; Niemelä 1999), the *Ecological value* method considered each habitat as an assemblage of smaller units with their own characteristics and values (Domon et al. 1986).

Few studies tested the accuracy of standardised habitat classification methods in representing insect communities (i.e. Rykken et al. 1997; Panzer and Schwartz 1998; Blake et al. 2003). Nevertheless, many authors argued for the use of insects, the most diverse living organisms group, as a biological indicator in doing an ecological assessment of habitats (e.g. Rosenberg et al. 1986; Kremen et al. 1993; McIntyre 2000; Taylor and Doran 2001). Among insect

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Table 1 Criteria used in calculation of ecological value indices (between 0 and 25)

Criteria	Definition
Uniqueness	Rareness and uniqueness estimation of the vegetation community of a given region
Representativeness	Correspondence (%) of floral composition with a typical community using classical studies as reference
Degree of succession	Inversely expressed by dominance of heliophilous plant species (heliophilous coefficient comprised between 0–1)
Specific richness	Total number of plant species
Rarity	Total number of rare species ^a

Each criterion has a given value between 0 and 5; see Cogliastro et al. (1996) and Boivin et al. (2003, 2005) for details on ecological value calculation

^a From the list of susceptible, vulnerable, and threatened vascular plants of Quebec province (Labrecque and Lajoie 2002)

groups, carabid beetles (Coleoptera: Carabidae) are widely and successfully used as indicators of disturbance intensity and as measure of biodiversity (Rainio and Niemelä 2003). Therefore, our objective was to determine if the *Ecological value* method is relevant to assess the composition and diversity of the carabid fauna. We used a previous study which applied the *Ecological value* method (Boivin et al. 2003) to investigate effects of vegetation characteristics (e.g. succession stage, layer structure, plant richness, etc.) on carabid assemblages. In fact, compositions of carabid assemblages are influenced by habitat type (Levesque et al. 1976; Niemelä and Halme 1992; Pearce et al. 2003), habitat heterogeneity and micro-climatic conditions (Liebherr and Mahar 1979; Epstein and Kulman 1990; Niemelä and Spence 1994; Niemelä et al. 1996; Magura et al. 2001).

Compared with forests, nearly null indices are attributed by the *Ecological value* method to open habitats, such as fallows or tall grasses (Domon et al. 1986; Cogliastro et al. 1996; Boivin et al. 2003). However, open habitats are known to hold more diverse carabid communities than forest habitats (Niemelä and Halme 1992; Heliölä et al. 2001; Magura et al. 2001). Consequently, we hypothesized that *Ecological value* would not represent the real value of urban open habitats concerning carabid richness and associated carabid species. Within forest habitats, mature forests are known to hold a greater carabid diversity than young ones (Southwood et al. 1979; Liebherr and Mahar 1979; Terrell-Nield 1990; Niemelä et al. 1996; Watts and Gibbs 2002). It is also generally admitted that an increased complexity of vegetation structure can positively influence diversity of animal fauna (Tews et al. 2004), notably diversity of carabid assemblages in forests (Liebherr and Mahar 1979; Niemelä et al. 1996). Therefore, we

hypothesized that the ecological value index of forest stations, and the vegetation criteria used in their calculation, would be correlated with richness, abundance and diversity of forest carabids. We focused on native carabid species because preservation and restoration of local indigenous fauna is an essential factor to consider in the conservation planning of urban environments (McKinney 2006).

Materials and methods

Study site

This study was conducted at Mount Royal conservation park (45°30 N, 73°36 W). The mount (elevation 232 m) is one of the Monteregians of southern Quebec, Canada. Surrounded by urban landscape, Mount Royal Park (190 ha) is a relic of temperate forest located in the urban center of Montreal. Our study was held on the north side of the hill, below the second highest summit (215 m), a property of the Université de Montréal (59 ha). The original forest has experienced intensive transformation since the development of the University infrastructures in 1924. The remaining vegetation (15.6 ha) is fragmented and is represented by a mosaic of hundred-years-old forests (e.g. red oak forest) standing next to young, recently disturbed ones (e.g. poplar forest and tall grasses) (Boivin et al. 2003, 2005). Poplar forest and brome tall grasses habitats were created by an old ski center and mixed tall grass habitat was created at the edge of an old snow disposal site (Fig. 1).

Habitats and stations selection

Eight habitats were chosen for sampling (Fig. 1). Within each habitat, Boivin et al. (2003) established permanent sampling stations of 20 m by 20 m in 2002 and 2003. In these stations, measurement and description of vegetation were previously taken (see methodology description in Domon et al. 1986; Cogliastro et al. 1996) in order to calculate ecological values indices (see Boivin et al. 2003, 2005). The ecological value indices were calculated using five criteria (Uniqueness, Representativeness, Degree of succession, Plant specific richness and Plant specific rarity) which have each a value between 0 and 5 (Table 1). Therefore, the ecological value indices have a value between 0 and 25. Twenty one forest stations and ten open habitat stations, which have specific ecological value index (Appendix A), were sampled (Table 2).

Carabid sampling

Carabids were collected using two pitfall traps in each permanent station placed 10 m away from each other in a

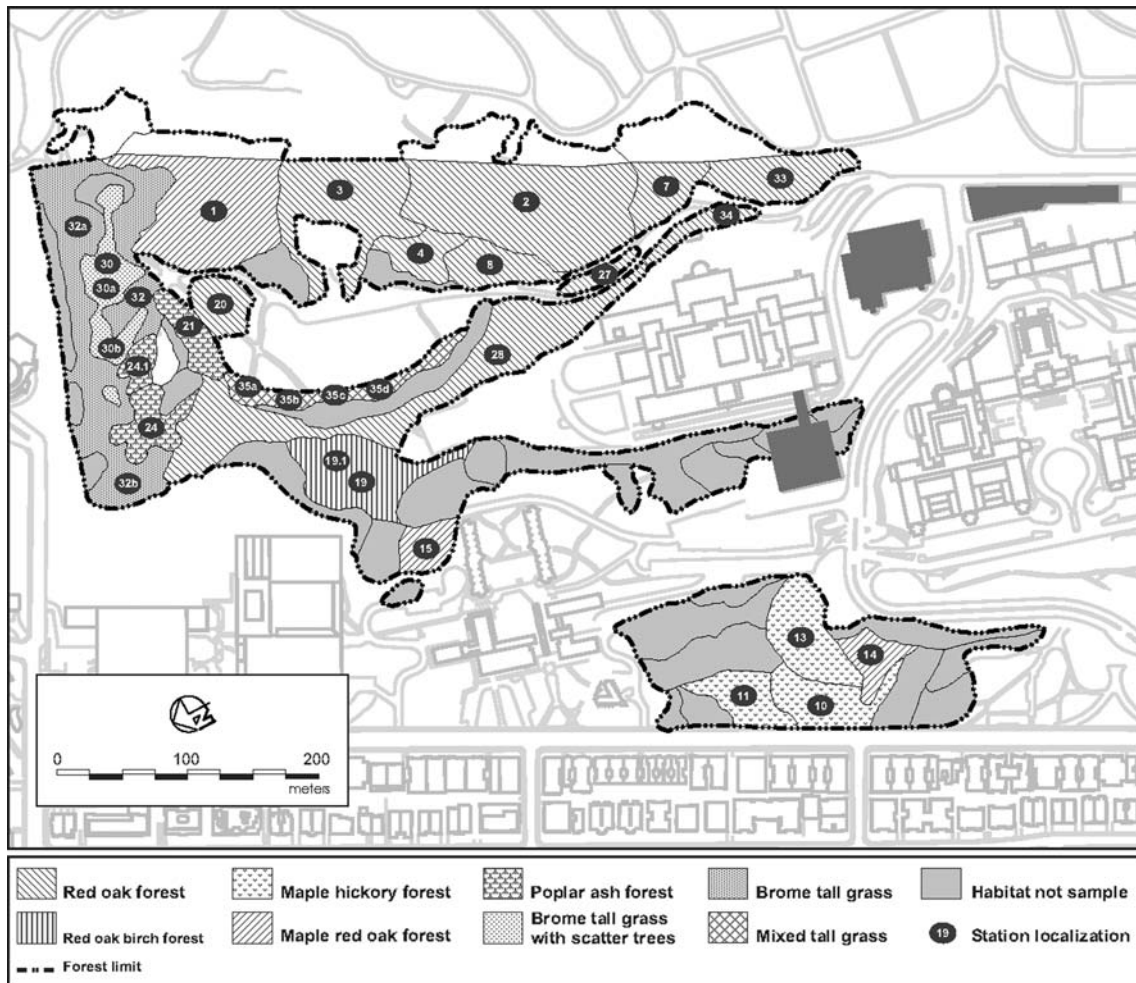


Fig. 1 Study site with habitat type and station localization

Table 2 Surface area of each habitat type sampled and number of stations used

	Habitats	Area (ha)	Number of stations
Forest habitats	Red-oak forest (<i>Quercus rubra</i>)	4.7	10
	Maple-hickory forest (<i>Acer saccharum</i> and <i>Carya cordiformis</i>)	1.3	3
	Maple-red oak forest (<i>A. saccharum</i> and <i>Q. rubra</i>)	1.9	3
	Poplar-ash forest (<i>Populus deltoides</i> and <i>Fraxinus americana</i>)	0.6	3
	Red oak-birch forest (<i>Q. rubra</i> and <i>Betula papyrifera</i>)	0.5	2
Open habitats	Brome tall grass with scattered trees (<i>Bromus inermis</i> and <i>Q. rubra</i>)	0.4	3
	Brome tall grass (<i>B. inermis</i>)	1.3	3
	Mixed tall grass (diverse grass and shrub species)	0.36	4
Total		11.06	<i>n</i> = 31

diagonal line. Thus, we used six pitfall traps per habitat type, except in the Red oak forest (20 pitfalls), the Red oak-birch forest (four pitfalls) and the Mixed tall grass (eight pitfalls) for a total of 62 pitfalls. Pitfall traps were 9 cm in diameter and were half filled with 100 ml of 1:2 water/propylene glycol. Plastic lids were placed ~2–3 cm over each trap to protect the traps from rain. Traps were placed

for seven consecutive days in June (from 18 to 25), July (from 15 to 22) and August (from 15 to 22) 2004. Due to disturbance by raccoons the June data was not used. Subsequent to high animal disturbance, traps were covered with wire mesh fixed to the ground with tent pegs. Carabid specimens were identified using the taxonomic reference of Larochelle (1976); Downie and Arnett (1996) and were

deposited in the Ouellet-Robert collection of the Université de Montréal (Quebec, Canada).

Environmental variables

Environmental variables used in the analysis of entomological data within forest stations were the ecological value of stations (varied between 0 and 25) and some of the criteria used in its calculation. We excluded the *Uniqueness* criterion because it corresponds exactly to habitat type and we didn't analyse the *Plant specific rarity* criterion because only one station had a rare species. Criterion of *Representativeness*, *Degree of succession* and *Plant specific richness*, were retained (Table 1). We have also added other variables susceptible to influence carabid assemblages. The age of forest stations (estimated using mean diameter, in centimetres, of the three largest tree trunks), the mean cover of shrub layer and the mean cover of herbaceous layer, were taken from the raw data of Boivin et al. (2003). Percentage cover of ground leaves was taken in the field doing a visual estimation within a 3 m diameter around each pitfall traps (Appendix A). Categories used in measure of shrub, herbaceous and leaves cover were: 1:<1%; 2:1–5%; 3:5–10%; 4:10–25%; 5:25–50%; 6:50–75%; 7:>75%.

Statistical analysis

Carabid species richness of different habitat types were compared using rarefaction curves. For this analysis, all stations of a particular habitat were pooled together. Individual-based rarefaction analysis gives an estimation of the expected number of species while taking into account sampling effort and total number of catches. By showing the rate of new species accumulation, rarefaction curves indicate if sufficient samples were collected to make proper comparisons of diversity (Gotelli and Colwell 2001). Thus, rarefactions are useful to compare sites in which sampling efforts were different (Buddle et al. 2005). Rarefaction curves were performed using the software program EcoSim700 (Acquired Intelligence Inc., Kesey-Bear) (Gotelli and Entsminger 2001).

To assess carabid associations with a particular habitat type, we used the Indicator Value procedure (IndVal) developed by Dufrêne and Legendre (1997). This approach combines abundance and occurrence of each species for various groups (here habitat type). A species which is found mostly in a habitat and is present in the majority of stations belonging to that habitat has a high IndVal. Low abundance sampled species (less than five individuals) can not be associated to a particular habitat type. The significance of IndVal (between 0 and 100) is obtained by a permutation test repeated 1000 times. This test was performed using the software program IndVal2.0. (Dufrêne 1998).

To investigate the effects of the ecological value index and other vegetation factors on carabid assemblages, we performed multiple comparisons within forest stations ($n = 21$). The entomological variables tested were: total abundance of carabids, abundance of native carabids, abundance of exotic carabids, total carabid richness, native carabid richness, exotic carabid richness and Simpson diversity index (Appendix A). We calculated correlations between our eight forest environmental variables and our seven entomological variables (matrix of fifty six correlations), as well as correlations among environmental variables (matrix of twenty eight correlations), using the non-parametric Spearman test because most variables were not normally distributed. To correct for spurious correlations, we applied an alpha inflation correction procedure called the False Discovery Rate (FDR), as proposed by Benjamini and Hochberg (1995). Indeed, calculation of a new restrictive probability level for each correlation (the FDR) eliminates the chances of erroneous positive correlations when performing several hundred simultaneous tests in ecological studies (Garcia 2004). The tests were executed using the software program JMP5.0.1 (SAS Institute 2002).

Results

Carabid assemblages

We collected a total of 658 carabid specimens. The carabid assemblage was composed of 32 species (21 native species, nine introduced and two undetermined *Amara* spp.) (Table 3). The two dominant carabid species, *Carabus nemoralis* and *Pterostichus melanarius*, introduced from Europe, composed more than 50 % of total carabids abundance. The eurytopic invasive species, *C. nemoralis* (37.5% of total abundance), was equally found in all habitat types, whereas *P. melanarius* individuals (16.1%) were collected mostly in mixed tallgrass. The two native species, *Diplocheila obtusa* and *Pterostichus mutus*, were also abundant and represented respectively 9.4% and 8.4% of the total number of carabid collected. *Diplocheila obtusa* was collected mostly in the mixed tallgrass and *P. mutus* was found to be eurytopic. *Trichotichnus autumnalis* represented a new mention to the province of Quebec (Bousquet pers. com.) and was deposited in the Canadian National Collection at Ottawa (Ontario, Canada).

Ecological value indices and open-forest habitat comparative analysis

Rarefaction curves of the Mixed tall grass and all forest habitats reached a plateau and so show sufficient sampling.

Table 3 Indicator value (IndVal) for each species in their preferred habitat

	Relative abundance	IndVal	Station presence
All habitats			
<i>Carabus nemoralis</i> O.F. Müller 1764 ^a	246/246	90.32 NA	27/31
<i>Pterostichus mutus</i> (Say 1823)	55/55	70.97 NA	21/31
Red oak forest			
<i>Chlaenius tricolor</i> Dejean 1826	3/3	30.00 NS	3/10
<i>Dyschiriodes globulosus</i> (Say 1823)	2/2	10.00 NS	1/10
<i>Harpalus plenalis</i> Casey 1914	1/1	10.00 NS	1/10
<i>Harpalus longicollis</i> LeConte 1848	1/1	10.00 NS	1/10
<i>Cymindis americanus</i> Dejean 1826	1/1	10.00 NS	1/10
<i>Anisodactylus harrisi</i> LeConte 1863	1/1	10.00 NS	1/10
<i>Amara</i> sp.2	1/1	10.00 NS	1/10
<i>Cymindis neglecta</i> Haldeman 1843	1/1	10.00 NS	1/10
Maple-hickory forest			
<i>Agonum melanarium</i> Dejean 1828	2/2	33.33 NS	1/3
<i>Clivina fossor</i> (Linné 1758) ^a	2/3	24.24 NS	1/3
Poplar-ash forest			
<i>Badister obtusus</i> LeConte 1878	2/4	33.33 NS	2/3
Red oak-birch forest			
<i>Amara lunicollis</i> Schiodte 1837	7/27	46.98 NS	2/2
Brome tall grass with scattered trees			
<i>Trichotichnus autumnalis</i> (Say 1823) ^b	1/1	33.33 NS	1/3
<i>Amara</i> sp.1	2/4	25.64 NS	1/3
Bromes tall grass			
<i>Ophonus rufibarbis</i> (Fabricius 1792) ^a	15/27	60.85*	3/3
<i>Amara musculus</i> (Say 1823)	1/1	33.33 NS	1/3
<i>Sphaeroderus lecontei</i> Dejean 1826	1/2	25.64 NS	1/3
<i>Amara aulica</i> (Panzer 1797) ^a	23/35	23.96 NS	1/3
Mixed tall grass			
<i>Poecilus lucublandus</i> (Say 1823)	18/18	100.00*	4/4
<i>Diplocheila obtusa</i> (LeConte 1848)	47/62	78.42*	4/4
<i>Pterostichus melanarius</i> (Illiger 1798) ^a	85/106	77.89*	4/4
<i>Harpalus rufipes</i> (DeGeer 1774) ^a	10/14	64.66*	3/4
<i>Bembidion quadrimaculatum</i> Say, 1823	6/6	50.00*	2/4
<i>Amara rubrica</i> Haldeman 1843	9/9	50.00*	2/4
<i>Harpalus affinis</i> (Schrank 1781) ^a	3/4	44.12 NS	2/4
<i>Amara bifrons</i> (Gyllenhal 1810) ^a	1/1	25.00 NS	1/4
<i>Amara apricaria</i> (Paykull 1790) ^a	1/2	17.86 NS	1/4
<i>Amara flebilis</i> (Casey 1918)	1/2	17.86 NS	1/4
<i>Harpalus pensylvanicus</i> (DeGeer 1774)	2/5	15.63 NS	1/4
<i>Harpalus fallax</i> LeConte 1859	1/4	11.36 NS	1/4

Relative abundance is represented by the number of specimens collected in this habitat over the total number of carabids collected. Station presence is represented by the number of stations in the preferred habitat in which the species was collected on the total number of stations used

* $P < 0.05$; NS: non significant; NA: non applicable

^a Introduced species (from Bousquet 1992)

^b New mention to the province of Quebec (Y. Bousquet pers. com.)

By contrast, rarefaction curves of both Brome tall grasses did not reach the asymptote and we can hypothesize that their richness was higher than shown. Rarefaction curves revealed, independently of sampling effort, higher expected carabid richness in the three open habitats than in all forest habitats except in the Red oak forest (Fig. 2a, b). Furthermore, the IndVal test indicated that seven species preferred open habitats while no species were associated

with forest habitat (Table 3). *Amara rubrica*, *Bembidion quadrimaculatum*, *D. obtusa*, *Harpalus rufipes*, *Poecilus lucublandus* and *P. melanarius* were strongly associated with Mixed tall grass and *Ophonus rufibarbis* was a specialist of Brome tall grass. Therefore, the null ecological values indices of open habitats were not corroborated with their high carabids richness and associated native species.

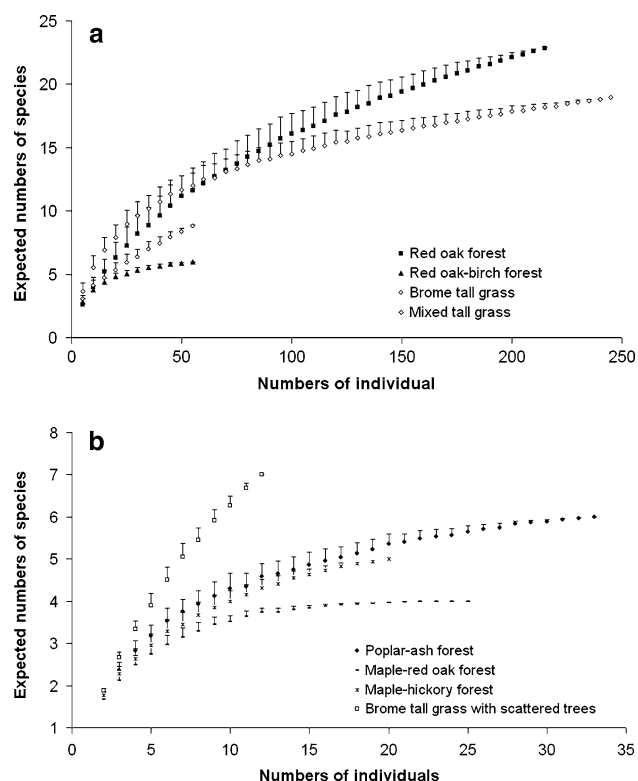


Fig. 2 Rarefaction estimates of expected number of carabid species, by sub-sample size (number of individuals) for total number of carabids collected within each habitat type **(a)** having a total number of individuals higher than 50 individuals; **(b)** having a total number of individuals less than 35 individuals). Error bars are ± 1 SD

Ecological value indices and forest variable correlations analysis

Ecological value indices of different forest stations were significantly correlated with native carabid abundance ($R_s = 0.57$; $P = 0.007$), but not significantly with native carabid richness ($R_s = 0.44$; $P = 0.048 > \text{FDR}$) and with other entomological variables ($P > 0.05$). The degree of forest succession, inversely expressed by heliophilous coefficient, showed significant correlations with total carabid richness ($R_s = -0.53$; $P = 0.013$) and native carabid abundance ($R_s = -0.59$; $P = 0.004$). Forest age, measured by tree diameters, was correlated only with the Simpson index of carabid diversity ($R_s = 0.60$; $P = 0.005$). Representativeness and plant specific richness criteria were not significantly correlated with any entomological variables (Table 4). Shrub and herbaceous cover were significantly correlated to total, native and exotic carabid abundance, as well as total and native carabid richness ($P < \text{FDR}$), whereas leaf cover were significantly correlated to none of the entomological variables ($P > 0.05$). Different relationships within forest stations' environmental variables are showed in Table 5.

Discussion

Relevance of the *ecological value* method for urban open habitats

The *Ecological value* method is not appropriate to represent carabid diversity of urban open habitats. Our results demonstrated that tall grasses hold rich carabid assemblages. According to rarefaction analysis (Fig. 2a, b), the three tall grass habitats (19, 9 and 7 species), had a significantly higher expected richness than forest habitats (max of 6 species). The only exception is the Red oak forest which is richer than all other habitat (23 species) probably due to its large surface area (i.e. it covers nearly half of the sampled area). An edge effect in the Mixed tall grass could be responsible for its great diversity since it is a small habitat which is next to forest. Indeed, carabid diversity can be increase by edge-associated carabid species (Magura et al. 2001) and we found four native carabid species to be associated to this tall grass, as established by their significant associated indicator values (see IndVal in Table 3). Habitat preference of those species is consistent with their known biology (Larochelle and Larivière 2003). Consequently, urban tall grasses hold rich carabid assemblages and associated native species while they have null ecological value indices. Moreover, Mount Royal spider and butterfly richness were also higher in tall grasses compare to forest habitats (unpublished data). Thus, we conclude that urban tall grasses play an important role in conservation of native carabid and most arthropods.

Relevance of the *ecological value* method for urban forests

The *Ecological value* method cannot represent effectively all carabid species in forests. Our results show that forest native carabids abundance is the only entomological variable significantly correlated with the ecological value index. In literature, other vegetation-based methods can represent, but only to a certain extent, general patterns of insect communities (Rykken et al. 1997; Panzer and Schwartz 1998; Blake et al. 2003). Indeed, some carabid species were collected exclusively in forest sites which did not necessarily have a high ecological value. For instance, two specimens of the native species *Agonum melanarium*, which live in hydrophilous habitats (Epstein and Kulman 1990), were only found in a Sugar maple-hickory forest site (station 11) because this site contains a temporary pool. However, this site is a disturbed forest station with a relative low ecological value index. With only vegetation criteria, we would be susceptible to lose this native species when doing potential future urban development.

Table 4 Spearman rank correlations (R_s) between entomological and environmental variables of forest stations ($n = 21$; $\alpha \leq \text{FDR}$)

	Carabid abundance			Carabid richness			Simpson index
	Total	Natives	Exotics	Total	Natives	Exotics	
Ecological value	0.36	0.57*	0.25	0.37	0.44*	0.10	0.18
Representativeness	0.30	0.48*	0.19	0.22	0.35	-0.11	-0.05
Heliophilous Coefficient	-0.44*	-0.59*	-0.31	-0.53*	-0.52*	-0.22	0.04
Plant richness	-0.16	-0.09	-0.12	-0.08	-0.11	0.10	0.45*
Mean tree diameter	-0.52*	-0.12	-0.52*	-0.12	-0.22	0.04	0.60*
Shrub cover	-0.75*	-0.64*	-0.67*	-0.59*	-0.65*	-0.12	0.17
Herbaceous cover	0.67*	0.58*	0.60*	0.65*	0.54*	0.41	0.23
Leaves cover	0.20	0.12	0.21	0.00	-0.06	0.15	-0.10

* $P < 0.05$; number in bold pass the False Discovery Rate

Table 5 Spearman rank correlations (R_s) between environmental variables of forest stations ($n = 21$; $\alpha \leq \text{FDR}$)

	Ecological value	Representativeness	Heliophilous coefficient	Mean tree diameter	Plant richness	Shrub cover	Herbaceous cover
Ecological value							
Representativeness	0.84*						
Heliophilous coef.	-0.59*	-0.54*					
Mean tree diameter	0.06	0.02	-0.16				
Plant richness	-0.01	-0.01	0.38	0.24			
Shrub cover	-0.43	-0.47*	0.43	0.44	0.24		
Herbaceous cover	0.26	0.20	-0.36	-0.10	0.32	-0.63*	
Leaves cover	0.55*	0.32	-0.22	-0.14	-0.31	-0.10	-0.16

* $P < 0.05$; number in bold pass the False Discovery Rate

We found that carabid richness (native and exotic added) and native carabids abundance were significantly higher in late succession forests (measured by their lower importance of heliophilous plant species). We also found a positive correlation between carabid diversity (expressed by Simpson index) and forest station age (measured by tree diameters). Our empirical results were consistent with Shea and Chesson’s (2002) prediction that more native species can coexist in mature community because they have more time to adapt to local conditions and to restrain their ecological niches. However, both estimations of forest maturity used (degree of succession and age of trees) did not show relationships with the same entomological variables and were not collinear. More experiments are needed in order to validate Shea and Chesson (2002)’s prediction in urban forest ecosystems.

An increase in plant diversity should lead to an increase in herbivorous insect diversity (Murdoch et al. 1972; Panzer and Schwartz 1998). This increase in prey availability has been hypothesized to result in higher carabid richness (Butterfield 1997; Liebherr and Mahar 1979; Hengeveld 1980). However, we found no relationship between carabids and plants richness. It is possible that

other factors such as vegetation structure, which also influence abundance and availability of prey, are more important in determining carabid community than the presence of particular plant species (Buse 1988; Niemelä et al. 1996; Brose 2003). In fact, carabid assemblages are known to be affected by herbaceous and shrub layer structure in forest areas (Niemelä and Spence 1994; Magura et al. 2001). On one side, closed forest habitats are known to hold very few carabid species (Butterfield et al. 1995; Niemelä et al. 1996), whereas on the other side an important herbaceous layer hold rich carabid assemblages (Butterfield 1997; Magura et al. 2001). Similarly, here, shrub layer cover had a negative and herbaceous cover had a positive relationship on abundance and richness of carabid assemblages. However, we cannot separate their individual effect because they were also negatively correlated with one another.

The *ecological value* method and urban biodiversity conservation

Animal taxa, such as insects, should be included in ecological classification methods, as complementary to vegetation

criteria (Refseth 1980; Eyre and Rushton 1989; Blake et al. 2003). The selection of a complementary faunal indicator depends on several factors, and one of them is the feasibility of sampling technique (McGeoch 1998; Rainio and Niemelä 2003). Our study showed that, in the urban regional context of the city of Montreal, it was very difficult to use pitfall traps due to scavengers as raccoons that made the collection effort too intensive. We conclude that the potential indicator insect taxa used as complementary to the *Ecological value* method have to be collected with another sampling technique. Furthermore, using a single faunal indicator group to complete vegetation data seems limited. It would be interesting to plan a sampling design (with another technique than pitfall traps) that allows measurement of different trophic guilds, including herbivores, decomposers, and saproxylic.

It is important and necessary to include arthropods in conservation strategy plans (Kim 1993), but there are among the most difficult taxa to cover satisfactorily (Redak 2000). The *Ecological value* method did not represent the high carabid richness of tall grasses and seems limited to represented entomological variables within forests. There could be different reasons as to why the *Ecological value* method does not successfully represent carabid diversity. First, the limiting sampling due to raccoon's disturbance problem could have resulted in a bias representation of the carabid community structure. Nonetheless, due to rarefaction curves results, we have confidence in comparison of the carabid diversity between open and forest habitats. When looking at stations within forest there is more doubt as if sampling was sufficient. For example, native carabid richness is not correlated with ecological value of forest stations ($R_s = 0.44$; $P = 0.048 > \text{FDR}$) but a more intensive sampling of carabid species could show a significant relationship. Second, it is also possible that, as carabid assemblages were numerically dominated by introduced species, biotic interactions (e.g. competition,

either exploitative or interference), especially between exotic and native species, could be more predominant than habitat factors in governing carabid assemblages. However, competition is not clearly demonstrated to drive carabid community organisation (Lövei and Sunderland 1996) and influence of exotic carabid species are not proven (e.g. Niemelä and Spence 1991; Niemelä et al. 1997). Finally, we raised some questions about the subjective choice of vegetation criteria and relevance of using a global floral index to cover the entire entomological fauna.

We show that at least some urban habitat characteristics are important to maintain faunal diversity and could be used in conservation planning. Indeed, our results highlight the importance of urban open habitats in conserving specific richness and associated native species of carabid assemblages. Furthermore, we underline interaction between maturity and structure of urban forest habitats and carabid assemblages. Knowing that urban ecosystems can harbour a considerable amount of biological diversity, notably in the case of insects (e.g. Eversham et al. 1996; McGeoch and Chown 1997; Eyre et al. 2003; Small et al. 2003; Watts and Larivière 2004), it seems important to look for an improved method of ecological evaluation to achieve better urban biodiversity conservation strategies.

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Appendix A Entomological variables for each forest and open station (July and August data was pooled) as well as criteria used in ecological value indices calculation, other environmental variables tested and habitat type (MRO: Maple-red oak forest; RO: Red oak forest; MH: Maple-hickory forest; ROB: Red oak-birch forest; PA: Poplar-ash forest; BT: Brome tall grass; BTS: Brome tall grass with scattered trees; MT: Mixed tall grass)

Station	Carabids abundance		Simpson index	Habitat type	Ecological value	Representativeness	Heliophilous coefficient	Plant richness	Mean tree diameter	Mean cover of shrub	Mean cover of herbaceous	Mean cover of leaves
	Native	Exotic										
1	2	8	2.78	MRO	12	33.3	0.110	28	58.3	4	4	7
2	5	10	2.42	RO	12	25	0.060	33	44.0	4	5	5
3	6	17	2.33	RO	11.5	25	0.050	16	39.9	3	5	5.5
4	3	3	3.60	RO	12.5	25	0.098	32	52.0	4.5	4	6
7	6	6	4.00	RO	16	50	0.006	34	59.8	3.5	7	4
8	2	16	1.59	RO	11.5	25	0.227	38	24.5	4	6	6
10	3	3	2.00	MH	10.5	38.5	0.162	43	50.9	4	5	1.5
11	4	6	4.55	MH	10	23.1	0.152	37	53.7	5.5	4	3
13	2	2	2.00	MH	12	33.1	0.071	30	52.1	4	4	5
14	1	1	2.00	MRO	11	33.1	0.072	29	44.5	5	3	3.5
15	4	9	2.32	MRO	12	33.1	0.031	28	42.8	5.5	4	5.5
19	9	13	3.46	ROB	13.5	50	0.077	38	34.2	3	6	5.5
19.1*	13	20	2.49	ROB	13.5	50	0.077	38	34.2	3	6	4.5
20	2	4	3.00	RO	11.5	25	0.240	37	45.5	4	4	4.5
21	2	18	3.65	PA	8.2	18.2	0.351	39	41.6	4	6	1.5
24	5	3	2.46	PA	8.2	18.2	0.367	28	32.9	4	4	4
24.1*	1	5	1.38	PA	8.2	18.2	0.367	28	32.9	4	4	2
27	6	39	1.46	RO	10.7	25	0.006	26	34.4	2.5	7	4
28	7	39	1.53	RO	13	50	0.058	28	31.5	3	5	5
33	2	13	1.32	RO	12.5	50	0.087	20	17.9	3.5	2	6
34	13	10	3.60	RO	13	50	0.015	28	-	3	6	4.5
30	2	3	4.45	BTS	0	-	-	-	-	-	-	-
30a	4	2	2.67	BTS	0	-	-	-	-	-	-	-
30b	1	0	1.00	BTS	0	-	-	-	-	-	-	-
32	0	4	1.00	BT	0	-	-	-	-	-	-	-
32a	3	31	2.05	BT	0	-	-	-	-	-	-	-
32b	2	17	3.03	BT	0	-	-	-	-	-	-	-
35a	15	37	4.31	MT	0	-	-	-	-	-	-	-
35b	37	21	3.99	MT	0	-	-	-	-	-	-	-
35c	42	61	5.46	MT	0	-	-	-	-	-	-	-
35d	11	23	4.28	MT	0	-	-	-	-	-	-	-

* Twin with the preceding station; a, b, c, d: stations created for this study

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