

The effects of urbanization on ant assemblages (Hymenoptera: Formicidae) associated with the Molson Nature Reserve, Quebec

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Abstract—Urbanization causes the fragmentation of natural habitats into isolated patches surrounded by anthropogenic habitats. Fragment size and the intensity of human disturbance have been shown to affect both composition and diversity of arthropod communities, but most groups have been understudied. We investigated effects of urbanization on ant assemblages (Hymenoptera: Formicidae) in and around the Molson Reserve, a preserved maple-beech forest surrounded by residential properties near Montréal, Quebec. We studied how local ant assemblages differed in terms of composition, abundance, and species richness, depending on whether they were situated in the interior forest, in adjacent residential backyards, or at the edge between these two habitats. We also compared an intact forest interior with a younger and moderately disturbed forest (“buffer zone”) between the urban matrix and the interior forest. Few differences were detected between the buffer zone and the intact forest interior. Extrapolated estimates of species richness suggest that it is lowest in the forest interior and highest in urban zones. Community composition, as investigated with ordination analysis, revealed a clear difference between the fauna of urban sites and the fauna of edges and forest interiors, and analyzing the relative abundance of ants showed residential backyards to contain the most ants. Urban assemblages were characterized by several competitively dominant species, including one introduced or “tramp” species. The occurrence of aggressive and dominant species in urban sites and at the edges of the Molson Reserve could potentially interfere with the dispersal and immigration of ground-dwelling arthropods and negatively affect local diversity or community composition in isolated forest reserves in urban centres.

Résumé—L’urbanisation fragmente les habitats naturels et les isole au sein d’espaces habités et modifiés par l’homme. Des études menées à l’échelle régionale suggèrent que la taille des îlots forestiers, ainsi que les perturbations humaines, affectent leur composition et la diversité des communautés d’arthropodes y habitant. Cependant, pour de nombreux autres groupes d’arthropodes ces effets n’ont pas encore été adéquatement testés. Nous avons exploré les effets de l’urbanisation sur les communautés de fourmis (Hymemoptera : Formicidae) à l’intérieur et aux alentours de réserves Molson, une érablière à hêtre adjacente à des propriétés résidentielles près de Montréal, Québec. Nous avons étudié comment l’assemblage des communautés de fourmis diffère en terme de composition et de richesse spécifique, en fonction de leur localisation, soit à l’intérieur de la forêt, de terrains résidentiels adjacents ou dans l’écotone séparant ces deux habitats. Nous avons aussi comparé le centre d’une forêt vierge avec celle d’une forêt plus jeune modérément perturbée qui tient le rôle d’un gradient plus traditionnel (« zone tampon ») entre la matrice urbaine et l’intérieur de la forêt. Bien que peu de différence ne fusse détectée entre la forêt vierge et la zone tampon, l’extrapolation de la richesse spécifique suggère que l’intérieur de la forêt est dotée d’une richesse spécifique plus basse, tandis que la zone urbaine comprend une richesse spécifique plus élevée. Comparativement à l’intérieur et à la lisière des forêts, les fourmis sont plus fréquemment collectées dans les terrains résidentiels, et sont composées de plusieurs espèces dominatrices, incluant des espèces introduites ou vagabondes. L’occurrence d’espèces de fourmis agressives et dominantes dans les sites urbains et dans les lisières (écotones) de la réserve Molson pourrait potentiellement interférer avec la dispersion et l’immigration des arthropodes terrestres, et affecter négativement la diversité locale ou la composition des communautés au sein des réserves forestières isolées dans les centres urbains.

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Introduction

The human population is increasing rapidly (Matson 1990), and as urban development intensifies, natural landscape features and properties are altered. In northern regions, urban ecosystems are typically fragmented landscapes where green spaces are surrounded by residential and business zones, industrial development, roads, and sidewalks. These urban features may reduce the dispersal efficiency of organisms including ground-dwelling arthropods (Niemelä *et al.* 2000, 2002). Analogous to fragmented natural forests, urban forests consist of isolated patches of vegetation with reduced area and, consequently, a greater proportion of edges (Halme and Niemelä 1993). Anthropogenic habitats are also characterized by impervious surfaces (pavement, asphalt), warmer temperatures due to the urban "heat-island" effect (Kim 1992), and higher numbers of invasive species (McIntyre *et al.* 2001).

It is important to document how urbanization may be altering species assemblages of arthropods, since these changes may affect ecosystem processes such as organic matter decomposition, soil aeration, and nutrient cycling (Bolger *et al.* 2000; McIntyre *et al.* 2001). This is especially relevant for ants (Hymenoptera: Formicidae), since ants are considered to be ecosystem engineers (Folgarait 1998), and they directly or indirectly affect resource availability for other organisms through nest building and waste management (Hart and Ratnieks 2002). They also increase the rate of nutrient cycling and stimulate decomposition (Folgarait 1998). Ants are excellent bioindicators of environmental change because their small size and reliance on high temperatures makes them especially sensitive to modifications of climatic and microclimatic conditions (Andersen 1997a; Kaspari and Majer 2000). Ants are long-lived and sedentary, providing highly reliable results from snapshot studies because occurrence patterns of ant species in a sampling grid are relatively constant through time (Alonso and Agosti 2000).

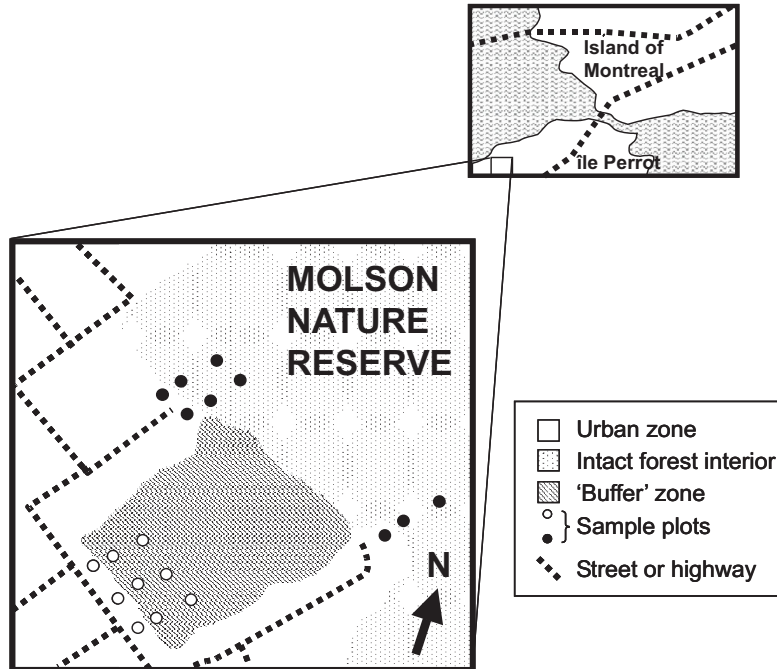
The key features of an urban landscape that can potentially affect ant assemblages in residential forests are patch size and edge effects. Gibb and Hochuli (2002) studied effects of habitat fragmentation on arthropod assemblages (including ants) in an urban environment in Australia. They found that arthropod assemblages in urban woodlands show a strong compositional

response to fragmentation, with small and large woodlands containing different assemblages of spiders and ants (Gibb and Hochuli 2002). They did not, however, detect significant differences in the species richness of arthropods among woodland fragments of various sizes. In a study near Québec (Québec, Canada), Francoeur (1966) showed that ant species richness was highest in disturbed stands in transition zones between forested and more urban areas. Dauber and Wolters (2004) studied edge effects in ant communities at ecotones between different types of agricultural landscape. Neither species richness nor nest density differed between the interior and the edge of different land use types. The relative abundance of some ant species, however, did vary with edge type, resulting in some compositional shifts in these communities. For example, dominant and aggressive species preferentially nested at edges (Dauber and Wolters 2004). This finding was supported by Bolger *et al.* (2000), who found the dominant and invasive Argentine ant, *Linepithema humile* (Mayr), to be more common at edges of scrub habitat patches in an urban system.

According to Forsys *et al.* (2002), urbanization promotes the establishment and invasion of nonindigenous species. Urban developments provide habitats with fewer native predators and competitors for an invasive species. This is particularly true for introduced ants (Forsys *et al.* 2002), as species such as *Solenopsis invicta* Buren and *L. humile* have been shown to have an affinity to urban environments (Carpintero *et al.* 2004). Urban areas also act as dispersal corridors for invasive ants, which in turn can increase the rate of invasion into adjacent and less disturbed habitats. The consequences for the native fauna may be severe, as documented in California (Suarez *et al.* 1998; Sanders *et al.* 2003). We are not immune to these potential effects in Canada, as invasive species have been documented in places such as the Mont St. Hilaire Biosphere Reserve in Quebec (Shik *et al.* 2005) and within the city centre of various Canadian cities — a topic currently in the public eye (*e.g.*, Staples 2004).

The objectives of the present study are the following: (i) characterize local differences in ant community composition in an urban forest and adjacent residential backyards; (ii) assess differences in ant assemblages among the interior forest, residential backyards, and the edge between these habitats; and (iii) test whether

Fig. 1. Location of the Molson Nature Reserve (Île Perrot, Quebec) and location of sampling plots (dark circles represent plots adjacent to or in the intact forest interior; white circles represent plots adjacent to or in the buffer zone). Residential properties line the streets directly adjacent to the forest and buffer zones.



ant species richness varies between an abrupt and a more progressive type of forest–urban transitional gradient.

Methods

Site description

The study area is situated in a residential suburb on Île Perrot, southwest of the Island of Montreal, Quebec ($45^{\circ}23'41.79''N$, $73^{\circ}58'32.45''W$) (Fig. 1). It includes a residential area and a conservation area, the Molson Reserve. This reserve, acquired by McGill University in 1998, is about 63 ha in area. It consists of forest, woodland marshes, swamps, and wetlands. The Molson Reserve is delimited by a lake on the north side, by train tracks to the east, and by houses and disturbed grasslands to the west and south (Fig. 1).

An old-growth forest represents the core of the Molson Reserve. This forest is on a bed of till and sand left by the last glaciation. It is composed mainly of deciduous trees including sugar maple, *Acer saccharum* Marsh. (Aceraceae), and American beech, *Fagus grandifolia* Ehrh. (Fagaceae), which are at least 80 years old. The Molson Reserve also includes

buffer zones or reserve extensions, which are adjacent to the southwest side of the old forest. These areas were partially cut several years ago for road development that was never completed and are now second-growth forest. These buffer zones show a high degree of disturbance compared with the central part of the reserve (e.g., slash, garbage, leaf piles, unofficial paths). Houses on the southwest side of the reserve were chosen for sampling based on the acceptance of owners and the degree of connectivity with the reserve (i.e., directly adjacent to the Molson Reserve).

Experimental design and sampling

A total of 18 sampling plots were chosen from the aforementioned habitats (Fig. 1). Six were in residential backyards: three of these were directly adjacent to the old-growth forest, and three were adjacent to the buffer zone (secondary forest). Six plots were positioned directly at edges (ecotones), adjacent to the aforementioned backyard sites. The six remaining plots were in the interior of the buffer zone (three plots) and the interior of the old-growth forest (three plots), perpendicular to the ecotone sampling plots. The urban plots were always

separated from the edge plots by 20 m, while the interior forest plots were 50 m from the edge plots. The distance separating the mini-transects was not predetermined but was a minimum of 50 m. The sampling area within each plot was a 2 m × 2 m grid (4 m²).

Pitfall traps — plastic containers used to collect surface-active arthropods (Bestelmeyer *et al.* 2002) — were dug into each corner of each sampling plot, with the upper trap edge level with the soil surface. The traps were plastic containers (7 cm × 9 cm) partly filled with a mixture of saturated saline solution and dish soap (one drop/trap). A plastic lid (10 cm diameter) was placed a few centimetres above each trap to prevent flooding by rainwater. Two 9-cm nails supported the circular lids. The traps were open for 3 weeks (16–30 June 2003). Samples were collected and sorted, and ants were preserved in 75% ethanol.

On 18 June 2003, 5 min of intensive visual surveys were done at each plot. All of the ants encountered were collected using an aspirator. When a nest was discovered, 10 workers from the same nest were collected, and a maximum of 5 winged specimens were collected when present. The ants were later preserved in 75% ethanol.

Baited traps were used to supplement visual surveys and pitfall trap collections. Baited traps were placed horizontally on the soil surface (Bestelmeyer *et al.* 2002). The traps were clear, cylindrical, plastic containers (10 cm height × 3 cm diameter). A trap containing honey (5 mL) was placed on the north side of the each sample plot and another with peanut butter (5 mL) was placed on the south side. Baiting was first done for a 4-h period (25 July) and later (14–15 August) for a 24-h period to capture night-foraging ant species.

Ants were mounted and subsequently identified to species, with verifications provided by A. Francoeur (Université du Québec à Chicoutimi). Voucher specimens are deposited in the Lyman Entomological Museum (McGill University, Macdonald Campus, Sainte-Anne-de-Bellevue, Quebec).

Data analysis

For each sampling plot, data from all four pitfall traps were pooled prior to analyses, as study questions were related to differences between plots rather than differences between individual pitfall traps. We used EstimateS (Colwell 2001) to generate species accumulation curves

from pitfall trap samples, using presence/absence (incidence) data. Species accumulation curves were calculated for each location (urban, edge, interior). This allowed for comparisons of accumulation rate among the different habitats sampled (Longino 2002). Curves reaching a plateau at the maximum sampling effort indicate that the observed species richness most likely accurately represented the actual species richness. However, because our sample sizes were low (*i.e.*, six per location), data were subsequently used to extrapolate an expected species richness had sampling continued. Three estimators (Chao-2, first-order Jackknife, and Bootstrap) were selected because they use incidence data and are widely used and accepted (Magurran 2004). The program EstimateS (Colwell 2001) was used to calculate these species richness estimates.

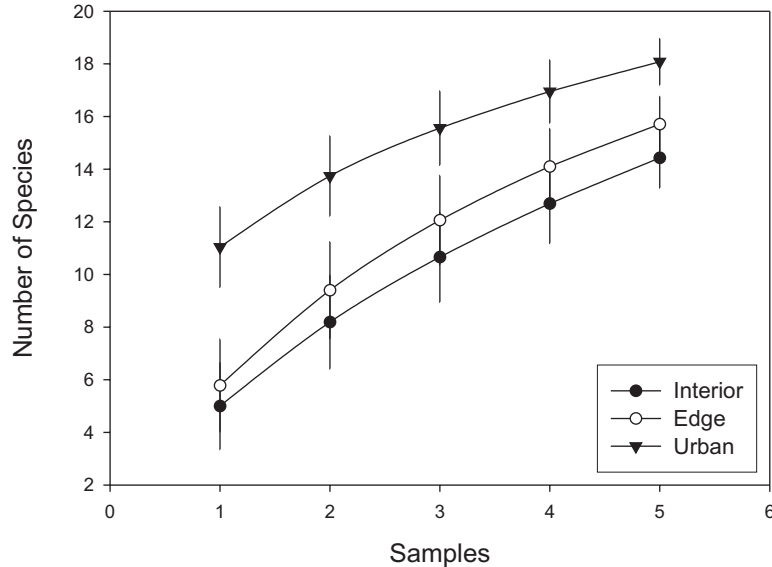
The relative abundance and species richness data were analyzed using two-factor analysis of variance (ANOVA), with trap location (urban, edge, interior) and type of transitional gradient (forest, buffer) as main effects. Data were transformed if assumptions of parametric statistics were violated. Pitfall trap sampling estimates the relative abundance of ants in the sampled locations (Bestelmeyer *et al.* 2002). Our analysis of abundance patterns was therefore restricted to pitfall trap catches. We further investigated species-specific responses for the most commonly collected ants. Common species were determined to be those collected more than 5% of the time. Since ant abundance may vary considerably depending on whether the pitfall trap is near or far from a nest, we used the percentage of plots (frequency) in which the commonly encountered ants were caught to determine occurrence patterns.

To assess whether the overall community composition of ants differed by gradient type or pitfall trap location, samples were subjected to a detrended correspondence analysis (DCA) ordination (Gaugh 1982). PC-ORD™ (version 4.0) was used for ordination analysis (McCune and Mefford 1999) of the presence/absence data recorded for each of the 18 sampling plots.

Results

Totals of 2196 individuals and 24 species were collected in and around the Molson Reserve (see Appendix A for complete species list). Pitfall trapping collected all of the species, whereas visual surveys and baiting yielded only

Fig. 2. Sample-based species accumulation curves for ants collected by pitfall traps placed in urban backyards adjacent to forests, at the ecotone (edge), and in the forest interior. Estimates are accumulated sample plots; although there were six sample plots per habitat type, only five are presented owing to the analytical resampling technique. Error bars are ± 1 SD.



small portions of the fauna. The total number of species caught in our experiment represents about 25% of the total number of ant species recorded in Quebec (A. Francoeur, personal communication).

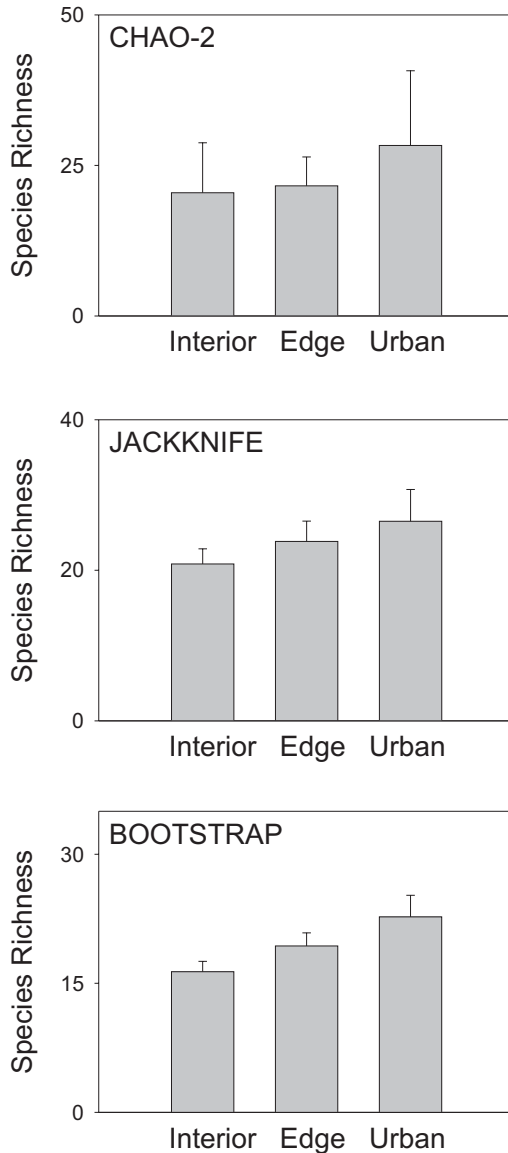
Species accumulation curves demonstrate that ant species richness accumulated rapidly as additional plots were sampled (Fig. 2). Although the curves begin to level, additional sampling would be required to completely inventory ant species of these habitats. There is a trend towards urban habitats containing the most species, followed by the ecotone; habitats located within the interior of forests had the lowest estimates of species richness (Fig. 2). Since sampling was probably insufficient, the use of estimated species richness is justified. The Chao-2 index was highly variable, and more sampling would be required to reduce the variance in this index (Fig. 3). The first-order Jackknife and Bootstrap estimates indicate that interior forest habitats harbour the lowest expected ant species richness, and urban habitats the highest expected species richness (Fig. 3), supporting the trend indicated by the species accumulation curves (Fig. 2).

Prior to analyses, species richness data were not transformed, but relative abundance data were log-transformed to conform to assumptions for parametric statistics. When analyzed

using a two-factor ANOVA, ant species richness was not significantly affected by either the location (urban, edge, interior) ($F_{2,12} = 1.377$, $P = 0.2894$) or the transitional gradient (forest or buffer) ($F_{2,12} = 1.061$, $P = 0.3232$), and the interaction term was nonsignificant ($F_{2,12} = 0.272$, $P = 0.7665$). The relative abundance of ants was significantly higher in urban locations (backyards) than in edge or interior locations ($F_{2,12} = 5.516$, $P = 0.020$) (Fig. 4). This was largely driven by frequent collections of *Formica subsericea* Say and *Lasius neoniger* Emery in urban samples (Table 1). The relative abundance of ants was not significantly affected by forest type ($F_{1,12} = 0.002$, $P = 0.962$), and the interaction term was nonsignificant ($F_{2,12} = 0.384$, $P = 0.689$).

Seven species were commonly collected (Table 1), and these can generally be classified into two groups. Four species (*F. subsericea*, *Formica glacialis* Wheeler, *L. neoniger*, and *Tetramorium caespitum* (L.)) were dominant in sampling plots located in backyards (urban habitat) (Table 1); although three of these were also collected in other habitats, the frequency of occurrence in edge and interior habitats was always generally lower than that in urban sampling plots (Table 1). *Tetramorium caespitum* was the only introduced species collected in this study.

Fig. 3. Estimates of expected ant species richness using three different estimators (Chao-2, first-order Jackknife, and Bootstrapping) for collections of ants in urban backyards adjacent to forests, at the ecotone (edge), and in the forest interior. Estimates are based on six sampling plots for each habitat type. Error bars are ± 1 SD.



The second grouping consisted of *Aphaenogaster* sp., *Camponotus pennsylvanicus* (DeGeer), and *Lasius alienus* (Foerster). These species were generally recorded as frequently in the interior forest as in at least one other habitat type (Table 1), which shows they were generally cosmopolitan in our study sites. *Aphaenogaster*

sp. could not be identified to the species level. We recorded *Stenamma schmitti* Wheeler in the intact forest interior and associated edges and residential backyards (Appendix A). This ant species is ubiquitous in more southern temperate forests (Gotelli and Ellison 2002) and was only once previously recorded in Quebec. Francoeur (1966) recorded the species for the first time while doing the inventory of the ant fauna in sugar maple stands in the region of the city of Québec.

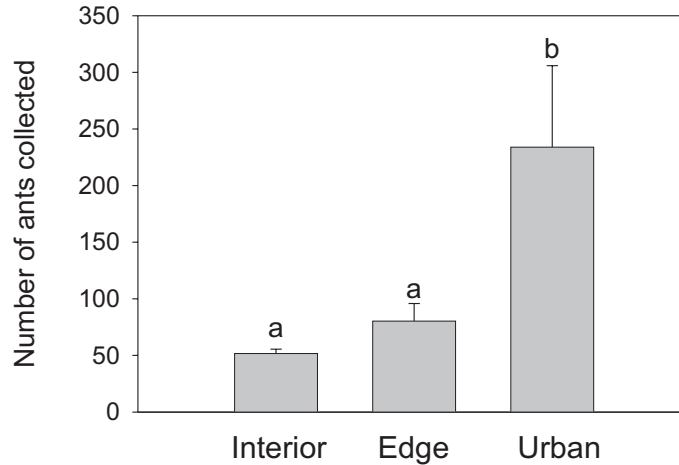
The DCA ordination of the 18 sample plots explained 53.6% of the variation in the species-sample matrix, and a three-dimensional solution is presented because Axis 3 contributes a substantial portion of this variation (Fig. 5). Axis 1 showed relatively little discrimination between the edge, interior, and urban sample plots, but Axes 2 and 3 (combined) suggested a general distinction between the ant assemblages collected in five of the six sample units placed in urban locations compared with edge and interior samples.

Discussion

Ant assemblages were different in residential backyards than in either interior forest or edge habitats, as evident by species-specific responses (Table 1), relative abundance of ants (Fig. 4), and overall community composition (Fig. 5). These results were partially due to the frequent occurrence of four open-habitat specialists in the urban zone: *L. neoniger*, *T. caespitum*, *F. subsericea*, and *F. glacialis*. In general, we found few differences in ant assemblages between our buffer-zone sample plots and the old-growth forest interior, suggesting that ants do not discriminate between these habitat types at the Molson Reserve. Accumulation of species richness suggests that additional sampling would likely be required to completely inventory the ant fauna in and around the Molson Reserve (Fig. 2). Extrapolated estimates of diversity (Fig. 3), however, suggest that interior forests harbour the least species-rich ant fauna and urban zones support the most species-rich fauna.

The unique aspects of the ant fauna in urban sites can be partially explained by species-specific patterns. *Tetramorium caespitum* and *L. neoniger* are both aggressive species that have high competitive ability (King and Phillips 1992). Although *L. neoniger* is native to North America, *T. caespitum* was introduced from

Fig. 4. Mean (\pm SE) number of ants collected (per sample, $N = 18$, $n = 6$) in urban backyards compared with the forest interior and ecotone (edge). Significant differences between means are indicated by different letters ($\alpha = 0.05$, Fisher's PLSD test).



Europe and is distributed worldwide. We can describe these species as “tramp ants” because they are closely associated with human activity, are geographically widespread, and have the potential to move into natural habitats (McGlynn 1999). These species are behaviorally and numerically dominant in habitats near human habitations and have the potential to affect local ant diversity through competitive or exploitative interactions (Sanders *et al.* 2003). Our study did not directly assess whether the presence of these species decreased species richness in locations where they were sampled, but nonetheless the catch patterns show that such tramp species dominate the urban ant fauna.

Formica subsericea and *F. glacialis* were also frequently collected in urban habitats. These species are typically associated with deciduous and mixed temperate forests (Francoeur 1973), but can also be collected in open habitats such as fields, early succession forests, and anthropogenic habitats (Francoeur 1973). These ants may benefit from urbanization because of the similarity of urban habitats to natural open habitats (*i.e.*, increased light availability and soil temperature) (Kim 1992) compared with forested habitats. In our study, *F. subsericea* was the most abundant ant species collected, and in the urban–forest transitional gradient, its relative abundance decreased from urban to forest habitats as the distance from residential backyards increased (Table 1). Interestingly, its occurrence was actually highest in the ecotone (*i.e.*, it was collected in four of six sample plots in the edge; Table 1), suggesting that

F. subsericea certainly can exploit the interface between an urban zone and forests, but it clearly avoids interior forest habitats. *Formica subsericea* and *F. glacialis* are also behaviorally dominant and may play a crucial role in shaping local ant community assemblages (Andersen 1997a). Our experiment thus strongly supports other research showing that highly competitive ants dominate in open habitats (Andersen 1997b; Bestelmeyer 2000).

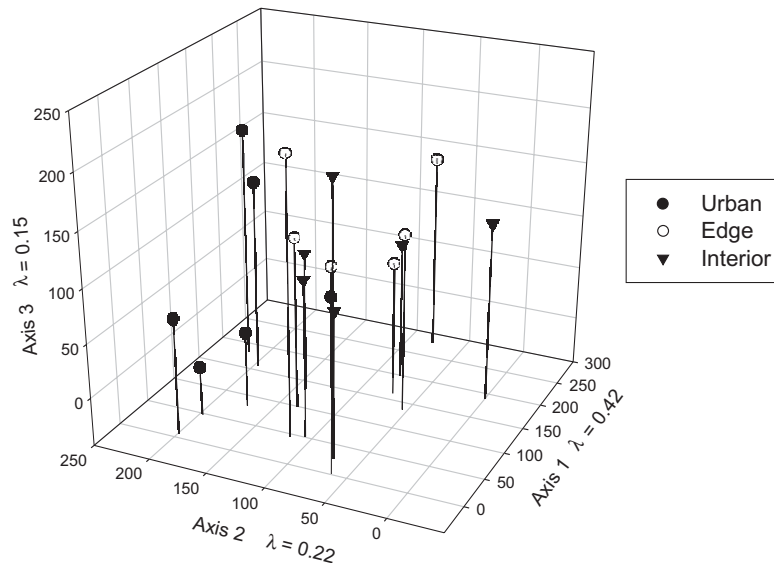
The ordination analysis showed some discrimination between urban ant assemblages and ant assemblages in the ecotones and forest habitats. Urban sites were inhabited by ant species that were practically absent from the edge and interior forest locations. The difference may be due to these species being restricted to urban sites or may be the expression of a divergence in the community assembly mechanism. It has been shown that the introduction of highly dominant ant species can disassemble native ant communities (Gotelli and Arnett 2000; Sanders *et al.* 2003). The peculiar community composition of urban sites may thus emerge from the exclusive presence of an introduced and dominant ant species such as *T. caespitum*.

The high expected ant species richness in urban sites (Fig. 3) is probably also explained by this mechanism. The presence of urban-zone specialists together with collections of the more cosmopolitan species (*e.g.*, *L. alienus*, *Aphaenogaster* sp., and *C. pennsylvanicus*) elevates expected richness in urban zones (Table 1, Appendix A). These species were sampled in most habitat types, including backyards. Forest

Table 1. Number collected and frequency of occurrence (%; in parentheses) for the most commonly collected species (>5% of the total catch) from pitfall traps placed in urban, edge, and interior habitats.

Species	Urban	Edge	Interior	Total
<i>Aphaenogaster</i> sp.	8 (50)	61 (67)	45 (67)	114 (61)
<i>Camponotus pennsylvanicus</i>	53 (83)	11 (67)	37 (67)	101 (72)
<i>Formica glacialis</i>	85 (67)	7 (33)	9 (33)	101 (44)
<i>Formica subsericea</i>	407 (33)	13 (67)	4 (33)	424 (44)
<i>Lasius alienus</i>	20 (50)	90 (100)	53 (83.3)	163 (78)
<i>Lasius neoniger</i>	317 (67)	33 (33)	1 (17)	351 (39)
<i>Tetramorium caespitum</i>	86 (50)	—	—	86 (17)

Note: Percentages represent the percentage of plots, for each habitat, in which a species was collected (six sampling plots, data pooled for all four pitfall traps within each sampling plot); total occurrence represents the percentage of all sampling plots (18 total) in which a species was collected.

Fig. 5. Detrended correspondence analysis ordination depicting ant assemblages (24 species) collected at sampling plots (18 total) placed in urban, interior, or edge sites. The species-sample matrix used presence/absence data.

ants may take advantage of the greater light availability in disturbed woodlands, edges, and residential backyards, whereas open-habitat specialists are confined to these habitats. *Aphaenogaster* sp. and *L. alienus* frequently occurred in old-growth forests; their populations may be linked to forest attributes such as high shade, increased moisture or humidity, and (or) decaying wood. They must also, however, possess competitive abilities that allow them to colonize more open habitats.

Combined with the presence of tramp species, the high abundance of dominant species in residential backyards could interfere with distributional processes operating between isolated

woodlands within the urban matrix. Urbanization facilitates the establishment of dominant, invasive ant species (Forys *et al.* 2002; Gibb and Hochuli 2003), and this may in turn have a peculiar effect if the urban matrix surrounds a patch of natural landscape. Dauber and Wolters (2004) found that dominant species of ants aggregated at edges between arable meadows. They suggested that high densities of nests of aggressive species with large colonies could reduce the permeability of undisturbed habitats to ground-dwelling arthropods. In other words, the presence of highly competitive species at habitat edges amplifies fragmentation effects by restricting access to the isolated patch by

potential colonizers. This may apply most appropriately to ants that are territorial and that compete through interference and aggression. In ant colonies, new queens are responsible for dispersal because after swarming and mating in a nuptial flight they typically go away from the natal nest and seek to found their own colony (Holldobler and Wilson 1990). Although new queens need to fly to be fertilized they must, at some point, land on the ground and prospect for suitable nest sites. Founder queens looking for nest sites are vulnerable to predation by workers of already established conspecific colonies, other ant species, or animals. This stage of a colony's life cycle has been shown to be a critical step in the process of successful dispersion and colonization (Fowler *et al.* 1986). High nest densities can thus increase mortality of incipient queens and prevent subsequent establishment of colonies. The Molson Reserve is almost completely surrounded by houses, and since each residential backyard sampled was dominated by either a single dominant species or a mix of these, it is possible that a decrease in species richness in the old-growth forest, as suggested by estimates of species richness, has been effected, or is in the process of being effected, by anthropogenic species interfering in the immigration processes of source populations. Future research and continued monitoring of ants at the Molson reserve is required to ensure the native fauna remains intact and to document additional effects of urbanization on arthropod biodiversity.

Acknowledgements

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Appendix

Appendix appears on the following page.

Appendix A

Table A1. Summary of ant species (presence or absence) collected by pitfall traps (72 total) placed in and around an old-growth forest and a closed-canopy buffer zone; traps were placed in urban sites (residential backyards), the interior of the forest or buffer, and at the ecotone (edge).

Species	Forest			Buffer		
	Urban	Edge	Interior	Urban	Edge	Interior
<i>Aphaenogaster</i> sp. Emery	*	*	*	*	*	*
<i>Camponotus nearcticus</i> Emery					*	*
<i>Camponotus pennsylvanicus</i> (DeGeer)	*	*	*	*	*	*
<i>Formica glacialis</i> Wheeler	*	*		*	*	*
<i>Formica neogagates</i> Viereck					*	
<i>Formica pergandei</i> Emery				*		
<i>Formica subsericea</i> Say	*	*	*			*
<i>Lasius alienus</i> (Foerster)	*	*	*	*	*	*
<i>Lasius nearcticus</i> Wheeler	*				*	
<i>Lasius neoniger</i> Emery	*	*		*		*
<i>Lasius pallitarsis</i> (Provancher)		*	*	*		
<i>Lasius umbratus</i> (Nylander)					*	
<i>Myrmica fracticornis</i> Forel				*	*	*
<i>Myrmica punctiventris</i> Roger		*	*	*	*	*
<i>Leptothorax longispinosus</i> Roger						*
<i>Myrmecina americana</i> Emery					*	
<i>Ponera pennsylvanica</i> Buckley	*					
<i>Solenopsis molesta</i> (Say)				*		
<i>Stenamma brevicorne</i> (Mayr)	*	*		*	*	*
<i>Stenamma diecki</i> Emery	*	*	*		*	*
<i>Stenamma impar</i> Forel	*	*	*		*	
<i>Stenamma schmitti</i> Wheeler	*	*	*			
<i>Tapinoma sessile</i> (Say)	*					
<i>Tetramorium caespitum</i> (L.)	*			*		