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# Ecosystem services provided by urban spontaneous vegetation

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**Abstract** Spontaneous vegetation colonizes large areas in and around cities. These unmanaged areas are considered to have low economic value or indicate dereliction, but recent research suggests that these can contribute valuable ecosystem services. This study evaluates indicators of ecosystem services in three habitats: urban spontaneous vegetation (USV), managed lawns, and semi-natural urban forest, in Halifax, Nova Scotia. USV had higher indicator values for habitat provisioning (plant species diversity, invertebrate abundance and taxonomic diversity) than the other habitats. Indicators of climatic regulatory services (albedo and leaf area index) in USV were similar to those in lawn habitats. Organic carbon content of the soils, an indicator of carbon storage, was lowest in USV but only marginally lower than in lawns. Standing biomass, an indicator of production services, was lowest in USV but lawn production may have been overestimated. While USV sites are usually transitory components of the urban landscape, they deserve further consideration due to their provision of ecosystem services, in some cases to a greater extent than conventionally valued urban habitats.

**Keywords** Pollinators · Habitat provisioning · Biomass · Species richness · Wasteland · Brownfield

## Introduction

Although ecosystem processes differ between human-dominated and natural environments, urban vegetation performs valuable ecosystem functions that benefit city inhabitants

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(Bolund and Hunhammar 1999). The role of urban forests, street trees, parks, and gardens in urban ecosystem functioning are well known (Nowak and Crane 2002; Akbari et al. 2001; Bolund and Hunhammar 1999; Freedman et al. 1996), but contributions to ecosystem functioning by other semi-natural areas, as well as created habitats like spontaneously colonized areas, are not well recognized. Ecosystem functions are the capacity of natural processes and components to provide goods and services that satisfy human needs (de Groot et al. 2002). Natural processes, such as decomposition, production of plant matter and nutrient cycling, are the result of complex interactions between biotic and abiotic components of ecosystems. Some recognize four primary groups of ecosystem functions: production, regulatory, habitat and information functions (de Groot et al. 2002). Each function is the result of natural processes of the ecological sub-system of which it is a part and each ecosystem process has associated goods and services. Ecosystem services are those functions considered to have value to people, either individuals or society (IPCC 2001). For simplicity, we will refer to ecosystem functions, goods and services together as ecosystem functions.

Production functions are a result of carbon fixation and provide ecosystem services such as food production, raw materials, genetic material, medicinal and ornamental resources. Biomass (dry weight) of above ground vegetation and abundance of invertebrates can serve as indicator variables for the provision of food or conversion of solar energy into plants and animals. Regulatory functions refer to the capacity of natural and semi-natural ecosystems to control essential ecological processes and life support systems through bio-geochemical cycles and other biospheric processes (de Groot et al. 2002). The advantages of urban trees and other plants in an urban setting include improved air quality, reduced air temperatures and lower energy demands for buildings (Akbari et al. 2001), as well as the absorption of gaseous pollutants (Nowak et al. 1998). The potential to positively affect air quality and air temperature is linked with the area of leaf surface (leaf area index or LAI) available for gas and water exchange and particle interception and reflectivity (albedo) of the vegetated area (Nowak 1994). Vegetation cover and ground surface temperature can also serve as indicators for the degree of surface shading.

Soils serve as the basis of many biogeochemical processes such as nutrient and water cycling and providing nutrients and habitat for soil fauna and flora (Bullock and Gregory 1991). Urban soils store carbon and intercept pollutants and other contaminants from human activities such as deicing salt (Cunningham et al. 2008). The contribution of urban soils as a carbon sink can be assessed using organic carbon content (Pouyat et al. 2006).

Pollination is a vital ecosystem function in terrestrial systems. Early successional wasteland vegetation can support a great diversity of pollinating insects due to the abundance of nectar producing flowering vegetation (Harrison and Davies 2002). Such habitats may also support nesting sites for ground nesting and wood nesting bees (Cane et al. 2006), which might be excluded from lawns and other urban habitats due to high levels of foot traffic and maintenance. The abundance and diversity of pollinators can serve as indicator variables for the support of regulatory pollinating services.

Urban habitats provide refuge, food and habitat for many plant and animal species, especially insects and birds (Gilbert 1989). Studies in the UK have shown high invertebrate richness and diversity in derelict and brownfield sites (see: Angold et al. 2006; Eyre et al. 2003; Small et al. 2003). These sites can provide conditions similar to natural habitats (such as sandy heaths and chalk grassland) and may help maintain populations of rare insect species (Eyre et al. 2003). In fact, some wasteland habitats associated with derelict and vacant land have received conservation status due to the presence of rare insect species (Harrison and Davies 2002). Plant species diversity (species richness, Shannon index, and native species richness) and invertebrate species diversity (species richness) can serve as indicator variables for habitat provisioning for these taxa.

Increasing interest in wasteland, brownfield, and other uncultivated vegetation is emerging in urban ecology. Many derelict, underused and abandoned spaces support vegetation that can be classified as 'spontaneous'—plants that colonize naturally without cultivation. These patches of urban spontaneous vegetation (USV) has also been referred to as 'urban commons' or 'urban wastelands' (Gilbert 1989), and ruderal vegetation (McKinney 2002). A variety of spontaneously colonized habitats (vacant lots, abandoned industrial areas, edges of parking lots, along rail lines, highways and other right-of-ways) frequently support a surprisingly high diversity of plant and animal species. In Europe, many USV habitats have been given considerable attention, including refuse tips (Darlington 1969), railway sites (Jehlik 1986), road verges (Klimeš 1987), wasteland (Sukopp et al. 1979), and old town centers (Brandes 1995) among others. In North America, urban ecological research has typically focused on remnant natural areas in cities rather than uniquely urban plant communities (Hope et al. 2003). Characteristics of urban vegetation in the Halifax area, Canada, have been investigated, including forests (Freedman et al. 1996; Turner et al. 2005) and spontaneous vegetation (Lundholm and Marlin 2006).

While USV is typically considered to have no or negative economic value, recent research suggests that there are many ecosystem services provided by such habitats. For example, brownfield land in Britain supports an estimated 12–15% of nationally scarce and rare invertebrates (Small et al. 2003). Findings like this stress the need to study areas of spontaneous vegetation colonization in and around cities.

An estimate from 2005, reports 61 130 ha of previously developed land (vacant land and buildings, derelict land, and for land currently in use and allocated for redevelopment) exists in England, with an estimate of 5.1% total land cover in London (English Partnerships 2006). A study by Simons (1998) of 31 cities in the United States estimated that between 5% and 10% of urban land is brownfield and that cities in the Northeast and Midwest states would have considerably more brownfield area due to their extensive industrial history. In a survey of 24 urban centers in Canada, respondents reported the percentage of urban area occupied by brownfield to be typically between 0.1% and 5% (De Sousa 2006), but no estimates were available for the Atlantic Provinces. Since the estimates for North America only include land that is contaminated or perceived to be contaminated, the percentages including uncontaminated derelict and underused properties would be much higher, and overall, land containing spontaneous urban vegetation makes up a considerable portion of land in and around urban areas.

In this study we quantify USV contributions to three groups of ecosystem functions (production, regulatory, habitat functions) by measuring indicator variables chosen to representing these functions at three common urban habitat types: urban spontaneous vegetation, lawn and remnant forest. We chose lawn and urban forest habitat for comparison as these are common in our region, relatively well studied, and considered highly desirable urban ecosystems by the public.

## Methods

### Study sites

The urban core of the Halifax Regional Municipality (HRM), which includes the former cities of Halifax and Dartmouth, has a population of over 280 000 and the highest population density in Atlantic Canada. The study area encompasses the Halifax peninsula and mainland area of urban Halifax and adjacent Dartmouth within Highway 111 (Online Resource 1). Pre-European settlement vegetation of the region is Acadian forest, which occurs within ecoclimatic zones considered cool temperate boreal (Weber and Flannigan 1997). The underlying geology consists of pyritic slate, schist, and migmatite rock types (AGS 1994) with podzolic, brown shaley loam soils (MacDougall and Cann 1963).

Examples of the following vegetation types in the study area were identified in 2007: (1) spontaneous vegetation, (2) lawns, and (3) remnant natural areas (forests). Sites were located by identifying possible areas using street maps and aerial photos (Google Earth) then chosen by ground surveys based on established criteria. Criteria for suitable USV sites were as follows: candidate sites had all original (natural) vegetation removed, with spontaneous colonization of vegetation and not actively maintained (to our knowledge). Eligible sites were at least 10 m×10 m with greater than 20% vegetation cover but less than 10% tree cover. Sampling sites were randomly chosen from a list of eligible sites based on ease of access (i.e. not fenced in or in areas with "No Trespassing" signs) and safety (i.e. no areas suspected to be polluted or with hazardous debris). For urban lawns, sites were at least 5 m× 5 m with greater than 90% vegetation cover and less than 10% by trees. Eligible lawns were actively maintained by mowing and had no ornamental shrubs or ground covers within the minimum size criteria. Urban forest sites were at minimum 10 m×10 m with at least 80% tree cover and not actively maintained. Forests and lawns were also chosen as close to the USV sites as possible. Twelve USV sites and five of each forest and lawn were established during the 2007 sampling season. However, one of the sites was lost to construction shortly after summer 2007; therefore, only 11 USV are included in the analyses.

The flora of the 11 urban spontaneous sites was documented by recording the plant species and cover in 12 1 $m^2$ plots per site. Five plots in each forest and lawn sites were sampled. We sampled fewer quadrats in forest and lawn sites because these sites were smaller, thus fewer quadrats were required to characterize sites, and fewer sites because the USV data is also being used for a more detailed study of vegetation characteristics to be reported separately. This plot size was chosen because it is considered optimal for non-treed vegetation (Krebs 1999). Plots were positioned using coordinates produced by a random number generator. The northeast corner was established at this point and the plot was oriented along a north–south axis. For ease in finding plots for the second field season via metal detector, each corner was marked by burying a metal washer just under the soil surface.

## Vegetation sampling

The point-intersect method (Krebs 1999) was used to estimate cover of species within the plots in 2007 during the antipated biomass peak (July–September). Each 1 $m^2$ plot was divided into 16 25 cm×25 cm subplots. A thin metal rod (1 mm diameter×1.5 m) was inserted at the intersection of a subplot within the plot, and all plant species contacted by the rod were recorded. Plants higher than the end of the rod were sampled by extending the rod vertically above the same point, and trees above 2 m tall were sampled by visually estimating height and percent cover above the entire 1 $m^2$ plot. Plant species were recorded only once per subplot. Point-intersect (PI) counts were used to generate plot-level summaries of total species richness, native species richness, total species abundance and native species abundance. The number of subplots with any vegetation contacting the rod was divided by 16 to estimate total percent cover of vegetation in each plot. In addition, if a species was present in the plot but not intersected by the rod it was recorded and included the total plot species richness and given a low value (0.25%) when intersect counts were tallied, so that they could be included in species diversity indices, but representing less cover than a single contact (1/16 or 6.25%). Species diversity indices (species richness, Shannon index, and native species richness) were calculated using PI counts. Species richness was determined as the cumulative number of species encountered within a plot. A detailed analysis of plant species composition is reported elsewhere (Robinson 2011), but in general, lawns were dominated by grasses, USV by tall forbs and grasses, and urban forest plots by trees, shrubs and lower cover of herbaceous species.

## Variables representing ecosystem functions

Variables representing ecosystem services and functions were sampled throughout the 2008 growing season (May-October). Species richness and Shannon-Wiener index were calculated using PI vegetation data. Substrate temperature, light availability at ground level and albedo (reflectivity of surface), were measured at each plot at midday (1100–1400 h) on a clear-sky day three times (once each in June, July and August). Temperature measurements were taken by placing a digital thermometer at the substrate surface three times over the sampling season (to a depth of 5 mm). Temperature readings were also taken at nearby paved surfaces; daily high air temperatures were taken from the Windsor Park weather station in central Halifax.

Light and albedo measurements were made in the center of the plot using a light meter (model number: LI-250A, Licor Biosciences). Light availability at ground level was measured using a quantum photometer (model number: LI-190SA, Licor Biosciences) that measures incoming photosynthetically active radiation (400 to 700 nm). To be able to compare surface shading for multiple days, light availability at surface was calculated as a ratio of unobstructed incoming radiation just above the tallest vegetation in the plot and light at ground surface. If vegetation was taller than 2 m, light and albedo reference (incoming) measurements were taken at approximately 1.5 from the ground in the closest not under the shade of the vegetation. Albedo was measured by taking upward and downward readings with a pyranometer sensor (model number: LI-200, Licor Biosciences) at a height of approximately 1 m. Albedo of the plot was calculated as the ratio of upward and downward values. Albedo was measured three times over the sampling season (June, July, and August) and an average of measurements is used in the analysis.

Above-ground plant biomass was sampled in mid-August by clipping all vegetation at ground level within a 10 cm strip oriented along a north–south axis centered in each plot. Plant material was placed in paper bags and oven-dried at 70°C for at least 48 h and weighed. For lawns biomass was estimated by multiplying the biomass of the clipping sample by the number of times (13) one of the lawns was mown during the growing season. Forest biomass was not sampled directly but references were obtained from several published studies of similar forest types (Botkin and Simpson 1990; Freedman et al. 1996).

Leaf area index (LAI) or one-sided green leaf area per unit ground area was calculated using a 20 randomly selected USV plots and 5 randomly selected lawn plots collected during biomass sampling. All leaves were scanned at 600 dpi on a flatbed scanner and leaf area calculated using Leaf Area Measurement software (version 1.3) (Askew 2003). Linear regression was performed to obtain a regression equation that would predict LAI based on plot cover ($LAI = 0.0072x + 0.7568$, where x = plot cover).

To determine organic carbon content, substrate samples of approximately 500 mL were taken from the center of the each plot. If surface covering prevented sampling from the center of the plot, samples were taken from as close to the center as possible. All samples were taken to a depth of approximately 10 cm, with the result that different soil layers were sampled in different habitats, as the upper layers in forest habitats are mainly organic layers, and USV habitats have rocky mineral soil. Organic matter content was determined by loss on ignition after 1 h at 450°C.

Sweep net samples were taken to capture plant feeding and resting invertebrates at each plot, three times during the May–October 2008 sampling period, not necessarily on the same days that other variables were measured. All samples were collected between the hours of 1000 and 1430 on a sunny day with little wind. Four sweeps were taken while proceeding in a line through the center of each plot, beginning and ending about 0.5 m from the edge of the plot. Because some plots were contiguous, during sweep netting insects may "flee" from a plot and be subsequently captured in an adjacent plot. To prevent this, non-adjacent plots were sampled before going back to sample the remaining plots. Samples were euthanized using killing jars containing ethyl acetate. Samples were stored in a freezer at −20°C until processing and identification several days later.

Ground-roving invertebrates were sampled using pitfall traps placed in the center of each plot. The pitfall traps were unbaited, consisting of plastic cups (65 mm diameter, 250 mL volume) containing approximately 50 mL of 75% ethylene glycol as a killing/preserving solution. The traps were covered with linoleum/ceramic tiles, larger rocks or bark pieces to protect them from litter and rain. Trapped invertebrates were collected at 2 week intervals during the sampling period. Samples were washed and stored in alcohol until processing and identification. For analysis the samples were pooled from the 5 month period.

All adult invertebrates were identified to species if possible (individuals were assigned to a morphospecies if identification past family was not possible) and then tallied by family or morphogroup (usually order or sub-order). Identification was facilitated by the use of insect collections at the Nova Scotia Natural History Museum in Halifax, Nova Scotia and the Nova Scotia Department of Natural Resources in Shubenacadie, NS, as well as the expertise of Dr. Christopher Majka, research associate of the Nova Scotia Museum and J. Scott McIvor, PhD. Candidate, Biology Department, York University, Toronto, ON.

Insect guilds regarded as important pollinators (bees (Hymenoptera and Apoidea); some wasps (Hymenoptera, Vespidae); flower flies (Diptera, Syrphidae); Bee Flies (Diptera, Bombyliidae) and butterflies (Lepidoptera, Papilionoidea and Hesperioidea)) were tallied and diversity (species richness) and abundance of these taxa were generated for each plot.

Two-sample t-tests were performed (R, Version 6.12; R Foundation for Statistical Computing, Vienna, Austria) on the variables representing ecosystem functions to determine statistical differences between habitat types.

## Results

All indicator variables differed significantly between habitat types (Table 1). USV plots differ significantly in all variables from the other two habitats, and lawns and forests differ significantly in all but four indicator variables (plant species diversity, vegetation cover, invertebrate family richness, and invertebrate abundance). Indicators of habitat provisioning, including plant species richness and diversity, and invertebrate richness and abundance were highest in USV compared to the other habitats. Light penetration and LAI were intermediate in USV, with forest having the lowest values of light penetration and highest LAI. Albedo was slightly higher in USV compared with lawns. USV had the lowest values for soil organic carbon, above-ground biomass, cover and highest temperatures (indicating the least contribution to microclimatic cooling). Measured on the same days, pavement temperatures

**Table 1** Mean and standard error for variables representing ecosystem functions for each of the three urban habitats sampled

| Variable | Spontaneous | Lawn | Forest |
|---|---|---|---|
| Plant species richness (# species/m$^2$) | 14.7±3.1$^a$ | 7.5±2.4$^b$ | 4.9±2.6$^c$ |
| Plant species diversity (H') | 2.0±0.2$^a$ | 1.1±0.4$^b$ | 1.1±0.5$^b$ |
| Soil organic carbon (%) | 4.3±0.5$^a$ | 5.4±0.7$^b$ | 24.4±5.1$^c$ |
| Above-ground biomass (g/m$^2$) | 342.6±44.5$^a$ | 1564.1±117.3$^b$ | 4180–13,100* |
| Vegetation cover (%) | 70±10$^a$ | 100±10$^b$ | 100±10$^b$ |
| Surface temperature (°C) | 24.10±0.76$^a$ | 20.39±0.95$^b$ | 16.99±0.48$^c$ |
| Light (at surface) (μmol/s/m$^2$) | 479.96±82.57$^a$ | 1499.92±114.76$^b$ | 26.66±5.33$^c$ |
| Albedo (reflected/incoming radiation) | 0.22±0.01$^a$ | 0.19±0.01$^b$ | 0.15$^†$ |
| Leaf area index (m$^2$/m$^2$) | 3.0±0.1$^a$ | 1.3±0.0$^b$ | 3.5–6.9$^‡$ |
| Invertebrate richness (# morphogroups/m$^2$) | 12.4±0.8$^a$ | 9.3±1.1$^b$ | 8.4±1.3$^b$ |
| Invertebrate abundance (# individuals/m$^2$) | 217.9±47.7$^a$ | 123.3±76.9$^b$ | 58.6±21.2$^b$ |

Different letters indicate statistically significant differences at $\alpha$=0.05

*Estimates from Botkin and Simpson 1990; Freedman et al. 1996

† Barry and Chorley 1992

‡ Chen et al. 2002

ranged from 8°C to 20°C warmer than air temperatures, with an average of 35.8±1.6°C (air temperature daily high for days sampled: 20.5±0.9°C).

Insects guilds known as important pollinators were caught in both sweep net and pitfall sampling. Twenty-two species of bees and at least 21 species of Lepidoptera, including eight species of butterfly, were found at USV sites across all plots. Captured butterflies included *Vanessa virginiensis* (American Lady), *Coenonympha tullia* (Common Ringlet), and *Pieris rapae* (Cabbage White), whereas many others were sighted but not documented; no butterflies were visible at lawn or forest sites, inside or outside the plots. Bees caught included European Honey Bee (*Apis mellifera*), eight species of bumble bee (*Bombus spp.*), six species of sweat bee (Halictidae) among others. In addition, twelve species of flower fly (Syrphidae), nine species of Vespid wasp (Vespidae) and three species of bee fly (Bombyliidae) were collected. No insects from the pollinator guilds were found in forests or lawns. Additional flower-dependent insects were found exclusively at spontaneous vegetation plots (Online Resource 2-1 and 2-2).

We counted a total of 262 species and morphospecies of invertebrates representing 93 families and morphogroups in both the pitfall and sweep net sampling, the majority of which were insects (Online resource 2-1 and 2-2). Of the non-insect taxa, woodlice and pillbugs (Isopoda) were the most abundant, followed by millipedes (Polydesmidae and Julidae) (Online resource 2-1 and 2-2). Across all pitfall samples, a total of 34 919 individuals were sampled; 30 397 in spontaneous plots (132 plots), 3083 in lawns (25 plots) and 1439 in forests (25 plots). USV plots had significantly higher average values per plot for all abundance and richness measures (Table 2). For sweep net sampling only, a total of 508 individuals were found, with the majority in spontaneous plots. No invertebrates were caught during sweep net sampling at lawn plots and 23 individual specimens were caught across all forest plots mostly consisting of flies (Diptera) including; grass flies (Diptera, Chloropidae), long-legged flies (Diptera, Dolichopodidae), as well as parasitic wasps "(Hymenoptera, Ichneumonidae) and one firefly (Coleoptera, Lampyridae). Interestingly, a greater abundance of invertebrates were collected in lawns than in forests, but invertebrate diversity was greater in forests.

**Table 2** Mean invertebrate abundance and richness per plot and standard error for each of the three urban habitats sampled

| Variable | Spontaneous | Lawn | Forest |
|---|---|---|---|
| Sweep abundance (# individuals/m$^2$) | 3.38±0.55$^a$ | 0 | 1.08±0.54$^b$ |
| Pitfall abundance (# individuals/m$^2$) | 214.84±47.73$^a$ | 123.32±76.94$^b$ | 57.56±21.38$^c$ |
| Total abundance (# individuals/m$^2$) | 217.92±47.70$^a$ | 123.32±76.94$^b$ | 58.60±21.21$^c$ |
| Sweep richness (# morphogroups/m$^2$) | 2.57±0.16$^a$ | 0 | 0.84±0.21$^b$ |
| Pitfall richness (# morphogroups/m$^2$) | 10.02±0.35$^a$ | 9.08±0.76$^b$ | 7.84±0.51$^c$ |
| Total richness (# morphogroups/m$^2$) | 12.38±0.40$^a$ | 9.28±0.54$^b$ | 8.40±1.33$^c$ |

Different letters indicate statistically significant differences at $\alpha$=0.05

During pitfall sampling, a small ladybug, *Hyperaspis inflexa* Casey, was discovered for the first time in the Maritime Provinces at a USV site (HF). Little is known about the distribution of this genus in the Maritimes, as this discovery represents a range extension of roughly 600 km (closest records are reported from Québec and New Hampshire) (Majka and Robinson 2009).

## Discussion

Compared with lawns and urban forest, the USV sites showed higher levels of ecosystem service provision for indicators of habitat provision, both for plant species diversity and invertebrate diversity. Most of the common plant species, however, were not native (Robinson 2011), thus the value of this habitat type for plant conservation habitat is uncertain. On the other hand, cover of native species varied from 20% to 67% in USV, with an average of 37%, compared to lawns with 20% (Robinson 2011) and so, while the most common species were not native, these areas may still represent an important habitat for native plants. Similar results have been reported from other regions. One study (in Berlin) found that wasteland and gravel pit sites supported the most plant species of all urban habitat types investigated, including forests and parkland (Gödde et al. 1995). USV sites also supported a great variety of life forms and functional types including annuals, biennials, herbaceous perennials, shrubs, trees, nitrogen fixing plants (species in the Fabaceae), and species with nectar producing flowers (Robinson 2011). While vegetation structure and composition is likely related to site age, we were not able to obtain accurate estimates of the length of time since abandonment for any of the USV sites, but our criterion for less than 10% tree cover for USV site selection may have restricted the USV sites in this study to less than 20 years since abandonment. Future studies should determine the whether habitat provisioning for invertebrates differs across a successional gradient in USV.

The abundance and variety of substrates and flowering plants in USV appears to support a more diverse invertebrate assemblage than in the other habitats. The volume of invertebrates caught at USV sites was higher than at forests and lawns combined. The capacity of USV to support higher trophic organisms may be enhanced by the diversity of plant species and lack of maintenance (i.e. biomass removal), as well as greater light availability due to lack of shading by nearby buildings (Matteson and Langellotto 2010). USV sites are at least contributing greater invertebrate abundance than lawn sites, likely by providing greater food and habitat values. The diversity of foods and habitats (richness of plant species) could increase overall productivity of the invertebrate assemblages. Interestingly, some of the invertebrates sampled at USV sites, like some of the plants, are not native. The European Honey Bee (*Apis mellifera*), Cabbage White (*Pieris rapae*) and European Skipper (*Thymelicus lineola*) butterflies, some of the flower flies (*Eristalis arbustorum, Eristalis tenax* and *Syritta pipiens*) millipedes in the family Julidae and some of the snails and slugs are introduced species.

Forested habitats certainly contribute considerably to urban production functioning, providing over 20 times more biomass than the other habitat types sampled. Invertebrates are undoubtedly abundant in remnant forest patches in urban HRM; however sampling methods would not have captured the abundance of invertebrates that primarily utilize mature tree habitat of forested areas such as canopy dwellers and bark borers, and requires further sampling to make a valid comparison. Carabid diversity in wasteland systems has a significant relationship with vegetation structure: greatest diversity is found among early successional tall herb plants (Angold et al. 2006). Gilbert (1989) reported a greater diversity of Carabidae and Lepidoptera larvae species at 4 to 6 year old brick rubble sites compared with older sites (12 to 15 years). This is likely due to the prevalence of open unvegetated habitat and greater diversity of Lepidopteran food plants found on the younger sites.

The presence of important pollinating insect guilds exclusively at USV sites highlights their importance in supporting insect pollinator populations synchronously with gardens and other types of urban cultivated vegetation. Spontaneously vegetated wastelands, however, are known to support a higher diversity of butterflies than any other urban habitat. Gödde et al. (1995) recorded 15 species of butterfly in wasteland habitat, more than parkland (11 species), native woodland (6 species) and field habitats (2 species). This is likely due to the prevalence of nectar and pollen producing flowering plants at USV sites. USV likely also provides more opportunities for ground nesting bees (sunny, well-drained, and either bare or partly vegetated areas) and wood nesting bees (rotting wood, dead branches and unpruned shrubs) than cultivated and maintained areas.

The USV sites supported a diverse invertebrate community, including pollinating, pest-controlling, and detrivorous species, each contributing ecological, educational and economic value within in the urban landscape. These results are similar to a recent investigation of urban insect diversity on green roofs and adjacent lawn and gardens in Halifax where 294 species were found at ground level (MacIvor and Lundholm 2011), many of which were retrieved in this study. Forest and lawn sites were not pollinator-rich, most likely a result of relatively few flowering plants in the forest understory or mown grass lawns. However, it is likely that the lack of sweep-netted invertebrates caught at forest plots was due to a sub-optimal sampling method for pollinating insects in forests. The type of forest sampled had little understory vegetation which may also have contributed to low invertebrate abundance. Several bees were seen visiting patches of white clover at lawn sites. It is likely that bees and wasps are present in urban forest and lawn habitats but not in quantities seen at the flower-rich USV sites. Dead tree limbs and snags can provide nesting sites for solitary wood nesting bees and wasps. Urban trees can harbor significant invertebrate communities including leaf-mining insects (Diptera, Lepidoptera, Coleoptera, Hymenoptera) (Smith et al. 2006), wood-boring and bark beetles (Coleoptera) (Brockerhoff et al. 2006), among other common plant feeding insects such as treehoppers, cicadas, and aphids (Hemiptera), but sampling methods limited the collection of these groups. Since USV habitats are dynamic and may only last a few years before re-development (Strauss and Biedermann 2006), future work investigating landscape variables such as proximity to other sites and natural areas should be done to further understand the habitat value of USV for pollinators and other insects.

Aboveground biomass dry weight was the chosen indicator variable for production functions in the three urban habitat types. Spontaneous plots had significantly lower standing biomass than lawn plots ($342.61 \pm 22.68$ g/m$^2$ versus $1564.12 \pm 59.87$ g/m$^2$); however, the values for lawn habitat are likely overestimated, since we harvested lawn biomass from close to the soil surface (so as to match the methods for the USV plots), lower than a lawnmower would clip. The value for spontaneous plots represent standing biomass at time of sampling while that for lawn plots represent the sum of production over the whole growing season. Values for turf production are reported to be 300 g/m$^2$/yr by Milesi et al. (2004) which is much lower than the production estimate in this study. While it is expected that urban forests have the most standing biomass compared to other urban habitat types, USV had standing crop levels similar to that of old field vegetation (Wiegert and Evans 1964).

During plot sampling, a woman was observed gathering grape leaves for use as food at one of the USV sites. This shows that production values of USV sites may extend beyond what is represented by a standing biomass indicator alone. The presence of species in USV absent from lawns and forest suggests that USV may also represent resources for wild-crafting (the practice of harvesting uncultivated plants for food, medicinal, or other purposes). Traditional usages for urban-associated plant species are widely known. Lund (1974) noted that 75% of uncultivated urban plant species in central Atlanta, Georgia had some recorded ethnobotanical significance ranging from food, medicinal and horticultural use.

USV albedo and LAI were greater than in lawn habitat, and only slightly lower for soil organic carbon content, thus USV could provide significant regulation services as well. Albedo at urban spontaneous plots (0.216±0.009) was slightly higher than at lawns (0.194± 0.006). Reported summer albedo values tend to be lower in forests than lawn and USV values measured here (with a maximum of around 0.15 for coniferous stands (Barry and Chorley 1992)) and boreal forest types (Betts and Ball 1997), and up to 0.18 for deciduous forest (Barry and Chorley 1992). Important to note is that measured albedo differs depending on measurement method and instrument choice, thus we cannot compare our measured albedo values with reference values for forest habitat from the literature, but our lawn and USV values are consistent with generally accepted values (around 0.2) for grasslands, used for climate modeling (Houldcroft et al. 2009). While cover was lower than lawns in USV plots, higher LAI, and the presence of concrete and other light colored substrate materials may have contributed to our measured albedo values. Thus, USV has the potential to provide equivalent services in terms of urban microclimate regulation through reflection (and possibly evapotranspirative cooling, given the relatively high LAI values) compared with lawns and natural grassland habitats. While soil surface temperatures were higher in USV than in the other habitats sampled, temperatures were still 10°C lower on average than pavement habitats, thus USV still contributes a valuable cooling service in urban environments. It should be noted as well that many lawns sampled were shaded by street trees and could have been watered, whereas the USV sites were selected to have low tree cover and receive no intentional inputs, thus the climate modifying properties of lawns indicated by our temperature values may be caused by other factors in addition to the lawn vegetation.

The leaf surface area of USV indicates a considerable capacity to filter and trap air pollution. Spontaneous vegetation plot LAI was twice as high as lawns and near the low end of the values for Acadian forests. Treed and forested areas indisputably outperform other vegetation in the city, but unmown grasses and other herbaceous vegetation also make an important contribution toward improving air quality (Currie and Bass 2008). With regular mowing, lawns barely reach an LAI of 1.5 m$^2$/m$^2$ (Milesi et al. 2004), while uncut grass typically has twice as much leaf surface area (3 m$^2$/m$^2$) (Currie and Bass 2008). LAI in Acadian forest can range from 3.5 m$^2$/m$^2$ (in deciduous stands) to 6.9 m$^2$/m$^2$ (in conifer stands) (Chen et al. 2002).

Soils function as carbon sinks but degradation by human activities has greatly influenced their functioning and development in urbanized areas (Effland and Pouyat 1997). The heat island effect influences carbon storage of urban soils because higher ambient temperatures increase $CO_2$ production (respiration) (Emmett et al. 2004). Organic content was considerably higher in forest soils, because of the greater contribution of organic matter by leaves and fallen trees, and our sampling method, which was biased toward capture of upper layers of soil rich in organics in the forest plots. USV sites with high tree cover also tended to be high in organic carbon (Robinson 2011). On average, lawns had greater soil organic carbon than USV plots. This may be explained by increases in productivity due to the input of nutrients and water, and lower compaction (low physical disturbance) often encountered in low-density residential and institutional land use types (Lorenz and Lal 2009), as well as the greater abundance of rubble and rocks in USV soil. The longer growing season of cool season turf grasses also contributes to an increased soil carbon density (Pouyat et al. 2003).

Above ground carbon storage was not estimated for USV or lawns mostly because the amount of woody vegetation among sites was not consistent. However, some sites supported larger trees and/or had significant shrub cover. Mature urban forests have been shown to store comparable amounts of organic carbon to more natural areas, including forested parkland (Freedman et al. 1996). If woody vegetation were to mature in spontaneously colonized urban spaces, the capacity for carbon storage (and therefore pollution mitigation) would be increased.

## Conclusions

The value of ecosystem functions are often not addressed in urban planning and development decisions (CBIN 2005). Integrating the value of ecosystem services from all city habitats including USV in vacant areas and transport right-of-ways will support the understanding of a complex and dynamic urban landscape. Of all the variables representing ecosystem functions, species richness and abundance measures, as well as LAI and albedo provide the strongest evidence that USV contributes equivalently or greater to certain urban climate regulation processes and habitat provisioning compared with other urban habitat types. Patches of USV within the urban land cover matrix may have a significant moderating effect on local climactic conditions, at least compared to hard surfaces/built environments. Vegetation has significant microclimatic effects in cities, reducing summer temperatures by several degrees (Dimoudi and Nikolopoulou 2003). This beneficial effect on air temperature improves as area of vegetation increases but also as the ratio of vegetated to built area increases. Therefore, by tolerating the growth of USV city-wide, the cumulative effect of these unofficial green spaces may be realized. USV sites that support more woody species would be more effective in climate regulation functions because trees and shrubs filter out more air pollutants than herbaceous vegetation (Currie and Bass 2008). However, younger, earlier successional USV sites seem to support a greater diversity and abundance of plant and invertebrate species. While this study only estimates coarse indicators of a fraction of the total functions contributed by urban ecosystems, USV at least appears to provide a unique role in habitat provisioning for urban invertebrates.

Encouraging a variety of successional states of spontaneous vegetation (e.g. Strauss and Biedermann 2006; Kattwinkel et al. 2011) will contribute to a range of ecosystem services measured in this study. Since some of the variables representing ecosystem functions were comparable to those of forests and lawns, the presence of USV within the urban landscape should be seen as a compliment and enhancement of the urban quality of life. In order to understand the regional effects of the loss of USV sites due to redevelopment, urban planners and ecologists should pay attention to the spatial configuration and temporal dynamics of abandoned lots across the city. Ensuring a diversity of habitats within urban areas will improve the quality of urban environments.

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