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Bee Fauna and Floral Abundance Within Lawn-Dominated Suburban Yards in Springfield, MA

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Abstract

Private yards comprise a significant component of urban lands, with managed lawns representing the dominant land cover. Lawns blanket > 163,000 km^2 of the United States, and 50% of urban and suburban areas. When not treated with herbicides, lawns have the capacity to support a diversity of spontaneous (e.g., not planted) flowers, with the potential to provide nectar and pollen resources for pollinators such as native bees. In order to determine the extent to which suburban lawns support these important species, we surveyed lawns in 17 suburban yards in Springfield, MA, between May and September 2013 and 2014. Householders participating in the study did not apply chemical pesticides or herbicides to lawns for the duration of the study. We collected 5,331 individual bees, representing 111 species, and 29% of bee species reported for the state. The majority of species were native to North America (94.6%), nested in soil (73%), and solitary (48.6%). Species richness was lower for oligolectic (specialists on a single plant; 9.9%) and parasitic species (12.6%). Abundance percentages for number of individuals were similar. We documented 63 plant species in the lawns, the majority of which were not intentionally planted. The most abundant lawn flowers were dandelion (*Taraxacum officinale*) and clover (*Trifolium sp.*). Nearly 30% of the spontaneous plant species growing in the lawns were native to North America. Our study suggests that the spontaneous lawn flowers could be viewed as supplemental floral resources and support pollinators, thereby enhancing the value of urban green spaces.

Key words: suburban yard, urban ecology, biodiversity, lawn, bee conservation

The United States encompasses one of the most densely urbanized regions in the world, with 82% of the populace living in urban areas (United Nations Population Division 2014). Urbanization is reported to degrade, fragment, and isolate natural habitats, changing both abiotic and biotic ecosystem properties that impact wildlife (Czech et al. 2000, Alberti et al. 2003, Foley et al. 2005, Cane et al. 2006, Shochat et al. 2010, Jha and Kremen 2013). Bees are ecologically and economically important because they are keystone species, they provide important ecosystem services to humans in the form of crop pollination (Kearns et al. 1998), and they may be undergoing regional population declines (National Research Council 2007).

Urban bee populations may be compromised by pesticide exposure, pathogens and parasites, reduced or degraded nesting habitat, limited or poor quality floral resources, and climate change (Cane and Tepedino 2001, Potts et al. 2010, Burkle et al. 2013), all of which can be associated with the intensification of urbanization (Ahrne et al. 2009, Winfree et al. 2009, Threlfall et al. 2015). Conversely, the presence of green spaces such as community gardens and residential yards (Fetridge et al. 2008, Matteson et al. 2008) retain habitat features, such as floral rich patches, and exposed soil, that are documented to enable many bee species to persist in urban areas (e.g., Owen 1991, McIntyre and Hostetler 2001, Tommasi et al. 2004, Baldock et al. 2015), yet few specialists bees have been recorded in these habitats (Frankie et al. 2005). In addition to responding to local habitat features, bees are also influenced at the broader landscape scale. For example, smaller fragments of habitat embedded within the urban matrix tended to support a higher proportion of small-bodied bees compared to larger, native grassland habitat in Boulder Colorado (Hinners et al. 2012). Further, Hinners and colleagues (2012) concluded that the bees recorded in the urban matrix have access to both native and novel habitats, which boosted overall species richness, especially when the amount of native habitat was significant. Thus, the resources found within urban green spaces might supplement the resources provided by native habitat resources. With the continued expansion of urban and suburban developments, it becomes imperative to understand how habitat elements within these landscapes can support wild bees and other beneficial insects (Lowenstein et al. 2015). Because bees are key pollinators, ensuring their persistence in urban areas is important for retaining urban biodiversity by providing pollination services for

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plants (both native and ornamental), and subsequently food and mast for other organisms (Sheffield et al. 2003, Biesmeijer et al. 2006, National Research Council [NRC] 2007).

Reports of bee assemblages within different urban and suburban green spaces confirm and support the presence of bees in these spaces, though these habitats vary, with some studies reporting rich native bee communities while others highlight more depauperate systems. In a study assessing community gardens in New York City, Matteson and colleagues (2008) documented that 19% of the 54 bee species they collected were exotic. This may be partially explained by an abundance of exotic and cultivated plant varieties in urban settings (Thompson et al. 2004, Frankie et al. 2005). In contrast, a study from a New York suburb that focused on flower gardens embedded within yards documented that only 5% of the 110 species of bees collected in 21 yards were exotic (Fetridge et al. 2008). Further, they found that ecological characteristics of the suburban bee community were more closely aligned with bee communities from a nearby forest preserve rather than more urban sites (e.g., New York City) with regards to nesting substrate (65% nested in soils) and the presence of parasitic species (19%; Fetridge et al. 2008). In northwest Ohio, private yards with a higher percentage of native plants supported greater bee richness and abundance (Pardee and Philpott 2014). At a Kentucky turf grass research station and in public parks, Larson and colleagues (2014) documented a rich assemblage of pollinators, including 37 bee species visiting dandelions and white clover, both of which were growing spontaneously in turf and park lawns. In two northern California cities, Frankie et al. (2005) surveyed bees visiting ornamental plants in residential gardens and reported that sites with a rich diversity of bee-attracting plants supported the greatest diversity and abundance of bees. These and other studies indicate that floral and nesting resources found within urban green spaces can support bee communities in urban areas. However, the ubiquitous lawn within yards has largely been ignored regarding its capacity to support urban bee abundance and diversity (Fetridge et al. 2008). Thus, a better understanding of this landcover's potential may provide the foundation for additional recommendations and guidelines to support urban bee conservation and management.

Residential yards and gardens (hereafter yards) comprise a large percentage of urban and suburban land cover and green spaces (40–50%; Nowak et al. 2001). Yards are generally dominated by lawn cover (the focus of our research), though often include tended gardens ranging from flower and vegetable patches to foundational shrubs and trees. In fact, lawns cover >164,000 km^2 of the United States, roughly 2% of all US land cover (Milesi et al. 2005). Landscaping choices such as the use of native or exotic plants, the presence and extent of lawns, and the use of pesticides can have implications for the wildlife inhabiting these yards (Gels et al. 2002, Goddard et al. 2010). For example, yards landscaped with native plants attracted more native lepidopteran larvae and insectivorous compared with yards landscaped with exotic plants (Burghardt et al. 2009). In Phoenix, AZ, lawn-dominated yards encouraged invasive birds whereas yards landscaped with desert plants supported higher abundances of native birds (Lerman and Warren 2011). Since bee abundance is closely related to pollen and nectar afforded by flowering plants (Frankie et al. 2005), and a large percentage of private yards consist of lawns, landscaping behaviors such as lawn mowing practices and the application of herbicides and pesticides likely have implications for the weedy floral diversity in the lawns. This could further implicate bee habitat quality (Fetridge et al. 2008).

Many environmental organizations encourage gardening practices that invite pollinators to public and private spaces (Mader et al. 2011), such as the National Pollinator Garden Network (NPGN), a collaboration of stakeholders from the garden, pollinator and conservation communities working together to support the health of pollinating animals. One of NPGN's major initiatives is the sponsoring of the Million Pollinator Garden Challenge (http://millionpollinatorgardens.org). These efforts have gained much recognition and attention, including the White House with their unprecedented National Strategy to Promote the Health of Honey Bees and Other Pollinators (Pollinator Health Task Force 2015), and represent a "call to action" to enrich bee habitat. Other conservation and wildlife organizations recommend the removal of certain plants deemed unworthy for wildlife habitat. For example, the Cornell Lab of Ornithology's YardMap program (http://yardmap.org) recommends reducing lawns because of the reputation of chemically treated lawns as nonhabitat for wildlife (Bormann et al. 2001, Tallamy 2007). However, to the best of our knowledge, the habitat value of lawns has yet to be evaluated. Given the large extent of lawns, this land cover requires increased scientific attention to better understand their habitat values for bees in urban and suburban settings.

The goal of our study was to characterize bee communities within suburban lawns to gauge the value of this habitat to bee conservation. Specifically, we 1) quantify bee abundance and species composition in suburban lawns to facilitate comparison of our results with prior research; 2) summarize the ecological characteristics of bees using untreated lawns, including nesting habitat, sociality, and floral specificity, because the response of bees to urban green spaces is known to be associated with natural history characteristics such as floral specificity and nesting ecology (Fetridge et al. 2008). The association of bees and lawns may be affected by the flight distance capabilities of bees and the extent to which they range over other habitats (Cane 2001, Zurbuchen et al. 2010). Accordingly, we 3) analyze body size of bees in suburban lawns to estimate foraging distance following Cane (1987). Finally, because nectar and pollen resources are essential to bee survivorship and reproduction, and are known to strongly influence their distribution (Potts et al. 2003), we 4) record the spontaneous floral resources (i.e., weeds) in the untreated suburban lawns. Although land conversion in the United States from urban to suburban has declined since the housing boom of the 1990s, between 2007 and 2010, urban and suburban areas have increased by 2.1%, and land developed between 1992 and 2010 has exceeded 10.3 million hectares (USDA 2010). Thus, a better understanding of the habitat resources in suburban yards might help to alleviate the detrimental impact of habitat loss. Practices that support nesting and foraging opportunities for bees could have important implications for bee conservation in suburban areas.

Materials and Methods

Study Area

We conducted the study in 17 yards in Springfield (Hampden County), MA, the third largest city in Massachusetts. Due to the somewhat invasive nature of sampling in private yards, we relied on volunteer households. We recruited households via ReGreen Springfield, a local tree planting organization. Yards were categorized as medium-density residential landuse and embedded within a suburban matrix. Participants owned and occupied single-family housing units. The yards encompassed a range of habitat features but were predominantly composed of lawns, although some sported limited flower borders or hedges, and none contained vegetable gardens (Fig. 1). Yards were not treated with herbicides or watered for the duration of the study, and were representative of Springfield yards. Participating yard parcel size ranged from 0.03 to 0.18 ha

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(typical of medium-density housing stock within Springfield), with a total of 1.26 ha sampled (Table 1). Canopy cover within a 50-m radius (centered at the center of each parcel) ranged between 2 and 49%, with a mean of 21% (Lerman, unpublished data). To promote spatial independence, all sites were at least 500 m apart with the exception of two yards, which were across the street neighbors. We assessed the degree of spatial autocorrelation with Mantel tests, which supported their independence ($r = 0.14$, $P = 0.13$).



Fig. 1. Examples of the lawn-dominated yards participating in the study from Springfield, MA.

Bee Sampling Methods

We collected bees between May and September 2013 and 2014, with a maximum of six sampling rounds per yard per year. Bees were collected from each yard approximately every three weeks (all collection rounds completed within 4 d of each other) on calm, sunny days using bee bowls and hand-held insect nets. Bee bowls consisted of white plastic 3.25 oz (96 ml) cups (Solo brand, model number p325w) painted florescent blue, yellow, or left white. In each lawn 30 bowls were placed in 10 arrays of three bowls, one of each color, and filled with a solution of soapy water (Dawn Ultra Dishwashing soap, original scent) to break the surface tension. Arrays were placed a minimum of 3 m apart near flowers growing in the lawn and in areas of the yard that received full sun for the majority of the day. We chose this method over a random transect method to standardize trap deployment in these small yards (0.03 to 0.18 ha) that were broken up by driveways, walkways, fences, and houses and other structures. Bowls were deployed for 24 h prior to a mowing event. Weather (cloud cover and temperature) was recorded. Bees collected in bowls were strained, and the specimens placed in whirl-paks with 70% ethanol alcohol and locality labels. The contents of each 30-bowl array were combined into one sample. We randomized the collection order for the yards amongst the different sampling rounds.

Because bowls tend to bias a collection toward smaller bees (Cane et al. 2000, Roulston 2000), we conducted 15-min hand-netting surveys in each yard for each sampling event concurrently with the bowl collection. We opportunistically hand-netted for bees on lawn and yard flowers. Upon capture, netted bees were placed in vials containing soapy water and then transferred into whirl-paks containing 70% ethanol alcohol. All bees were washed in soapy water, dried with a hair drier, pinned and labeled following LeBuhn et al. (2003). Bees were identified to the species level when possible using a number of different keys (Mitchell 1960, 1962; Gibbs 2010,

Table 1. Site characteristics, sampling effort, and bee and lawn flower biodiversity of 17 yards in Springfield, MA

Site ID	No. of sampling rounds	Total grass (m^2)	Ha	Bee species richness	Bee abundance ($n = 5{,}331$)	Lawn flower richness	Lawn flower abundance	Soil (%)	Canopy cover (%)
16_1	3	949	0.09	23	57	9	552	17.5	11.4
16_2	12	828	0.08	39	614	24	20,951	9.4	48.9
FP_1	6	262	0.03	21	87	3	1,733	3.7	29.7
EFP_1	6	349	0.03	39	329	13	16,986	27.9	1.0
EFP_2	5	456	0.05	22	103	15	15,591	0.0	0.2
16_3	12	793	0.08	47	305	21	23,089	3.8	13.0
EFP_4	6	324	0.03	38	215	8	492	7.1	13.7
EFP_5	5	489	0.05	21	99	18	12,365	3.0	6.0
16_4	6	1,805	0.18	41	300	19	3,223	6.8	39.7
EFP_6	12	825	0.08	46	494	21	10,907	4.7	3.5
EFP_7	12	830	0.08	45	489	38	18,597	9.7	5.6
EFP_8	6	756	0.08	27	75	9	165	5.0	41.1
FP_2	6	310	0.03	33	172	5	2,492	6.5	47.1
16_5	12	753	0.08	42	778	19	8,221	3.1	25.0
EFP_9	6	829	0.08	32	397	9	1,791	3.0	33.4
EFP_10	12	659	0.07	40	317	20	8,068	5.4	26.7
EFP_11	12	1,431	0.14	53	500	29	21,234	15.1	11.2

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2011; Gibbs et al. 2012). Vouchers are deposited with the US Forest Service Urban Natural Resources Institute.

Bee abundance and richness were calculated as the total number of individual bees and total number of species collected from both collection methods for the two years. Individuals not determined to the species level were omitted in the richness calculation but included in the total abundance. We calculated the proportion of each species based on the total number of bees collected per species and dividing by the total number of bees collected, as well as the frequency of occurrence across the sampled yards by determining the number of yards a species was collected and then dividing by the total number of yards sampled.

To facilitate easy comparison among urban bee studies, bees were classified with respect to ecological characteristics following Fetridge et al. (2008) and Matteson et al. (2008), based on the format provided by Giles and Ascher (2006). We compiled additional information from the primary literature, natural history accounts, and Discoverlife (Mitchell 1960, Hurd 1979, Michener 2007, Ascher and Pickering 2012). We classified bees based on their: 1) origin (native or exotic); 2) floral specialization (oligolectic or polylectic; oligolectics specialize on either a single plant species or family following Cane and Sipes (2006) whereas polylectics do not have a strict dietary preference); 3) nesting substrate (soil, cavity, soft or rotting wood, wood, or pithy stems); 4) sociality (solitary, subsocial, eusocial, or parasitic); and 5) body size (small, medium, and large). To estimate body size we measured the intertegular (IT) distance (distance between wing bases) (Cane 1987) on up to three individual females of each species. Based on these measurements, we averaged the intertegular distance for each species. This level of accuracy was adequate for the purposes of this study since we then grouped species into small (<1.5 mm), medium (1.6–3 mm), and large (>3.1 mm) size classes (Hinners et al. 2012). For species where specific ecological characteristic information was lacking, we inferred conditions based on closely related species. We calculated the percentage of ecological characteristics for these groups as the total number of individuals collected as well as total number of species to characterize the dominance of these different traits.

Flora Sampling

Prior to each collection event, we identified and estimated the total number of lawn flower blossoms in bloom to assess available floral resources. Lawn flowers were those growing spontaneously amidst the planted turf grass. We then classified the origin of each plant based on Lorenzi and Jeffery (1987). To quantify for the nonflowering plants, we conducted two intensive sampling events per yard, per year using the quadrat sampling method. The plots consisted of three 1-m^2 plots per yard whereby we identified every plant and bare soil, and assigned a percent coverage of that species / cover type for the plot. We included plants that we were unable to identify (i.e., "unknown 1," "unknown 2") to account for total floral abundance per lawn.

Results

Bee Fauna

We collected 5,331 bees belonging to 111 spp. (3,194 bee specimens consisting of 96 species in 2013, and 2,137 bee specimens from 82 spp. in 2014) in Springfield lawns. The majority of species (78%) were recorded from less than half of all the yards sampled. Species richness per yard ranged between 21 and 53 bees with a mean of 36 species (Table 1). From these specimens we document five New World bee families (Andrenidae, Apidae, Colletidae, Halictidae, and Megachilidae), 22 genera and 16 county records (Table 2), filling in gaps for the known distribution of bees in Massachusetts (J. Milam, personal observation. All but 18 specimens were identified to species: the exceptions were nine *Lasioglossum* (*Dialictus*) sp. males, eight *Ceratina* sp. in poor condition, and one *Nomada* sp. (female) recorded as *N. bidentate* because this taxonomic group is currently being revised (S. Droege, personal communication). We combined *Hylaeus affinis* and *H. modestus* observations because current keys cannot reliably separate them (Grundel et al. 2011). The most abundant species collected (17%) was *Lasioglossum illinoensis*, a bee previously thought to be at the northern edge of its range in southern Connecticut (Zarrillo et al. 2016), followed by *L. pilosum*, *Ceratina strenua*, *Halictus confusus*, and *H. ligatus* representing 36% of all individuals. Nearly three quarters of the collected species were represented by fewer than 10 individuals (Table 2), and 33 species were singletons. Species represented in the majority of the yards (>88%), but not collected in abundance included *Bombus impatiens*, *Agapostemon virescens*, *A. texanus*, *Apis mellifera*, *L. ephialtum*, *L. pectorale*, and *L. oenotherae*. Although the pan traps caught the majority of species (108 species compared with 33 species from the hand-netting), 7% of the specimens were captured using the 15-min hand netting survey method (Table 2).

We summarize the taxonomic and ecological characteristics of the bees collected from the suburban lawns in Table 3. The Andrenidae made up 17.1% of all species and 2.9% of all individuals. The Apidae were well represented with 22.5% of all species and 24.5% of all individuals collected belonging to this family. Bees from the Halictidae family were abundant with 44.1% of all species and 70.4% of all individuals. Few Megachilidae were captured (10.8% of the species and 1.4% of individuals; Table 3). Individuals in the family Colletidae were not well represented with 5.4% of species and only 0.8% of all individuals. Of the 111 bee species collected from Springfield yards, the majority of species and individuals were native to North America (94.6 and 94.1%, respectively). Six species were exotics that are well established in eastern North America (Cane 2003). The most common bee species comprised small- or medium-sized bees (45.0 and 43.2%, respectively) while small-bodied bees were most abundant (71.9%; Table 3)

The majority of the species and individuals were polylectic (76.6 and 96.5%, respectively) and soil nesters (73.0 and 73.9%). Almost three quarters of the bees collected were eusocial (69.8%) while nearly half of all species were solitary bees (48.6%; Table 3). We recorded 11 oligolectic species representing 147 individual specimens. Of note was the widespread *L. oenotherae*, recorded at 88% of all sites. We collected nine *Peponapis pruinosas*, from five yards. In addition, we captured six oligolectic *Andrena* spp. from five sites, a single *Colletes*, and two *Melissodes* (Table 2).

An eighth of the species collected were parasitic (bees that lay their eggs on or near pollen provisions collected by other bees), represented by two families, three genera, 14 species, and 38 individuals (Table 2). The occurrence of parasitic bees is dependent on the presence of their hosts. For the *Nomada* and *Sphecodes* bees, their associated hosts (see Sheffield et al. 2003) were present in all of the collection locations. The most abundant parasites were represented by the genus *Nomada*, which are parasitic primarily on the genus *Andrena*, but also of *Agapostemon*, *Halictus*, *Lasioglossum*, and *Colletes*, all genera represented in our study. We captured two species of parasitic *Lasioglossum* (*Dialictus*), *L. platyparium*, and one *L. izawsum* that are parasitic of pollen-collecting *Dialictus* spp., although we did not collect their presumed associated hosts (*L. katherineae* and *L. versatum*, respectively; Gibbs et al. 2012).

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Table 2. Bee species and ecological characteristics of bees collected in Springfield, MA, yards from 2013 and 2014

Species	Abund[a]	% Indiv[b] ($n = 5{,}331$)	% Pres[c] ($n = 17$)	Family	Orig[d]	Pollen[e]	Nest[f]	Behav[g]	Size[h]	Method[i]
Colletes inaequalis Say, 1837	22	0.41	59	Colletidae	N	P	S	S	M	B
***Colletes latitarsis* Robertson, 1891**	1	0.02	6	Colletidae	N	O	S	S	M	P
Colletes thoracicus Smith, 1853	1	0.02	6	Colletidae	N	P	S	S	L	H
***Hylaeus (Hylaeus) mesillae* (Cockerell, 1896)**	8	0.15	24	Colletidae	N	P	C	S	S	P
Hylaeus (Prosopis) affinis (Smith, 1853) / *modestus* Say, 1837	9	0.17	29	Colletidae	N	P	C	S	S	P
Hylaeus undet.	1	0.02	6	Colletidae	N	P	C	S	S	H
Agapostemon (Agapostemon) sericeus (Förster, 1771)	11	0.21	41	Halictidae	N	P	S	S	M	P
Agapostemon (Agapostemon) texanus Cresson, 1872	51	0.96	88	Halictidae	N	P	S	S	M	B
Agapostemon (Agapostemon) virescens (Fabricius, 1775)	175	3.28	100	Halictidae	N	P	S	S	M	B
Augochlora (Augochlora) pura (Say, 1837)	1	0.02	6	Halictidae	N	P	SW	S	M	P
Augochlorella aurata (Smith, 1853)	17	0.32	47	Halictidae	N	P	S	E	S	P
***Halictus (Nealictus) parallelus* Say, 1837**	5	0.09	24	Halictidae	N	P	S	E	M	B
Halictus (Odontalictus) ligatus Say, 1837	295	5.53	100	Halictidae	N	P	S	E	M	B
Halictus (Protohalictus) rubicundus (Christ, 1791)	26	0.49	76	Halictidae	N	P	S	E	M	B
Halictus (Seladonia) confusus Smith, 1853	385	7.22	100	Halictidae	N	P	S	E	S	B
Lasioglossum (Dialictus) anomalum (Robertson, 1892)	1	0.02	6	Halictidae	N	P	S	E	S	P
***Lasioglossum (Dialictus) bruneri* (Crawford, 1902)**	28	0.53	47	Halictidae	N	P	S	E	M	B
Lasioglossum (Dialictus) coreopsis (Robertson, 1902)	2	0.04	6	Halictidae	N	P	S	E	S	B
Lasioglossum (Dialictus) cressonii (Robertson, 1890)	18	0.34	53	Halictidae	N	P	SW	E	S	P
Lasioglossum (Dialictus) ellisiae (Sandhouse, 1924)	8	0.15	41	Halictidae	N	P	S	E	S	P
***Lasioglossum (Dialictus) ephialtum* Gibbs 2010**	208	3.90	100	Halictidae	N	P	S	E	S	P
Lasioglossum (Dialictus) fattigi (Mitchell, 1960)	1	0.02	6	Halictidae	N	P	S	E	S	P
***Lasioglossum (Dialictus) illinoense* (Robertson, 1892)**	929	17.43	94	Halictidae	N	P	S	E	S	P
***Lasioglossum (Dialictus) imitatum* (Smith, 1853)**	89	1.67	71	Halictidae	N	P	S	E	S	P
Lasioglossum (Dialictus) izawsum Gibbs, 2011	1	0.02	6	Halictidae	N	P	[S]	P	S	P
***Lasioglossum (Dialictus) katherineae* Gibbs, 2011**	1	0.02	6	Halictidae	N	P	S	E	S	P
***Lasioglossum (Dialictus) laevissimum* (Smith, 1853)**	3	0.06	18	Halictidae	N	P	S	E	S	P
Lasioglossum (Dialictus) leucocomum (Lovell, 1908)	44	0.83	59	Halictidae	N	P	S	E	S	P
Lasioglossum (Dialictus) lineatulum (Crawford, 1906)	5	0.09	29	Halictidae	N	P	S	E	S	P
Lasioglossum (Dialictus) oceanicum (Cockerell, 1916)	2	0.04	12	Halictidae	N	P	S	E	M	P
Lasioglossum (Dialictus) oblongum (Lovell, 1905)	1	0.02	6	Halictidae	N	P	SW	E	S	P
Lasioglossum (Dialictus) pilosum (Smith, 1853)	764	14.33	100	Halictidae	N	P	S	E	S	B
***Lasioglossum (Dialictus) planatum* (Lovell 1905)**	4	0.08	24	Halictidae	N	P	S	E	S	P
Lasioglossum (Dialictus) platyparium (Robertson 1895)	3	0.06	18	Halictidae	N	P	[S]	P	S	P
Lasioglossum (Dialictus) smilacinae (Robertson, 1899)	17	0.32	41	Halictidae	N	P	S	E	S	B
***Lasioglossum (Dialictus) subviridatum* (Cockerell, 1938)**	2	0.04	12	Halictidae	N	P	S	E	S	P
Lasioglossum (Dialictus) taylorae Gibbs, 2010	1	0.02	6	Halictidae	N	P	S	E	S	P
Lasioglossum (Dialictus) tegulare (Robertson, 1890)	264	4.95	94	Halictidae	N	P	S	E	S	P
Lasioglossum (Dialictus) undet.	12	0.23	6	Halictidae	N	P	–	–	–	P
Lasioglossum (Dialictus) versans (Lovell, 1905)	1	0.02	6	Halictidae	N	P	S	E	S	P
Lasioglossum (Dialictus) versatum (Robertson)	1	0.02	6	Halictidae	N	P	S	E	S	P
Lasioglossum (Dialictus) vierecki (Crawford, 1904)	26	0.49	24	Halictidae	N	P	S	E	S	P
***Lasioglossum (Dialictus) viridatum* (Lovell, 1905)**	2	0.04	12	Halictidae	N	P	S	E	S	P
Lasioglossum (Dialictus) weemsi (Mitchell 1960)	44	0.83	76	Halictidae	N	P	S	E	S	P
Lasioglossum (Evylaeus) cinctipes (Provancher, 1888)	2	0.04	12	Halictidae	N	P	S	E	M	P
Lasioglossum (Hemihalictus) foxii (Robertson, 1895)	1	0.02	6	Halictidae	N	P	S	S	S	P
Lasioglossum (Hemihalictus) nelumbonis (Robertson, 1890)	7	0.13	18	Halictidae	N	P	S	S	S	P
Lasioglossum (Hemihalictus) pectorale (Smith, 1853)	105	1.97	100	Halictidae	N	P	S	S	S	P
Lasioglossum (Lasioglossum) acuminatum McGinley, 1986	4	0.08	24	Halictidae	N	P	S	S	M	P
Lasioglossum (Lasioglossum) coriaceum (Smith, 1853)	10	0.19	29	Halictidae	N	P	S	S	M	P
Lasioglossum (Leuchalictus) leucozonium (Schrank, 1781)	35	0.66	71	Halictidae	E	P	S	S	M	B
Lasioglossum (Sphecodogastra) oenotherae (Stevens, 1920)	125	2.34	88	Halictidae	N	O	S	S	M	B
Sphecodes confertus Say, 1837	1	0.02	6	Halictidae	N	P	[S]	P	–	P
Sphecodes coronus Mitchell, 1956	1	0.02	6	Halictidae	N	P	[S]	P	S	P
Sphecodes fattigi Mitchell, 1956	1	0.02	6	Halictidae	N	P	[S]	P	S	P
Sphecodes illinoensis (Robertson, 1903)	6	0.11	24	Halictidae	N	P	[S]	P	S	P
Sphecodes mandibularis Cresson, 1872	5	0.09	18	Halictidae	N	P	[S]	P	S	P
Andrena (Andrena) carolina Viereck, 1909	1	0.02	6	Andrenidae	N	O	S	S	M	P
Andrena (Andrena) frigida Smith, 1853	1	0.02	6	Andrenidae	N	O	S	S	M	P
Andrena (Callandrena s.l.) asteris Robertson, 1891	1	0.02	6	Andrenidae	N	O	S	S	M	P
***Andrena (Callandrena s.l.) helianthi* Robertson, 1891**	1	0.02	6	Andrenidae	N	O	S	S	M	P
Andrena (Callandrena s.l.) placata Mitchell, 1960	1	0.02	6	Andrenidae	N	O	S	S	M	P
Andrena (Larandrena) miserabilis Cresson, 1872	21	0.39	59	Andrenidae	N	P	S	S	M	B
Andrena (Leucandrena) barbilabris (Kirby, 1802)	1	0.02	6	Andrenidae	N	P	S	S	M	P
Andrena (Melandrena) carlini Cockerell, 1901	10	0.19	41	Andrenidae	N	P	S	S	M	B
***Andrena (Melandrena) commoda* Smith, 1879**	1	0.02	6	Andrenidae	N	P	S	S	M	P
Andrena (Melandrena) regularis Malloch, 1917	3	0.06	12	Andrenidae	N	P	S	S	M	B
Andrena (Melandrena) vicina Smith, 1853	2	0.04	12	Andrenidae	N	P	S	S	M	P
Andrena (Plastandrena) crataegi Robertson, 1893	3	0.06	6	Andrenidae	N	P	S	S	M	P

(continued)

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Table 2. continued

Species	Abund[a]	% Indiv[b] ($n = 5{,}331$)	% Pres[c] ($n = 17$)	Family	Orig[d]	Pollen[e]	Nest[f]	Behav[g]	Size[h]	Method[i]
Andrena (Scrapteropsis) alleghaniensis Viereck, 1907	14	0.26	41	Andrenidae	N	P	S	S	M	B
Andrena (Scrapteropsis) imitatrix Cresson, 1872	3	0.06	12	Andrenidae	N	P	S	S	M	P
Andrena (Simandrena) nasonii Robertson, 1895	9	0.17	18	Andrenidae	N	P	S	S	S	P
Andrena (Taeniandrena) wilkella (Kirby, 1802)	3	0.06	18	Andrenidae	E	P	S	S	S	H
Andrena (Trachandrena) hippotes Robertson, 1895	1	0.02	6	Andrenidae	N	P	S	S	M	P
***Andrena (Tylandrena) erythrogaster* (Ashmead, 1890)**	3	0.06	6	Andrenidae	N	O	S	S	M	P
Calliopsis (Calliopsis) andreniformis Smith, 1853	78	1.46	71	Andrenidae	N	P	S	S	S	P
Anthidium (Anthidium) manicatum (Linnaeus, 1758)	8	0.15	29	Megachilidae	E	P	C	S	L	P
Anthidium (Proanthidium) oblongatum (Illiger, 1806)	9	0.17	41	Megachilidae	E	P	C	S	M	B
Hoplitis (Alcidamea) producta (Cresson, 1864)	10	0.19	41	Megachilidae	N	P	P	S	M	P
Megachile (Eutricharaea) rotundata (Fabricius, 1793)	5	0.09	24	Megachilidae	E	P	C	S	M	B
Megachile (Litomegachile) mendica Cresson, 1878	6	0.11	29	Megachilidae	N	P	C	S	M	B
Megachile (Megachile) centuncularis (Linnaeus, 1758)	5	0.09	24	Megachilidae	N	P	C	S	M	B
Megachile (Megachile) montivaga Cresson, 1878	2	0.04	12	Megachilidae	N	P	C	S	M	P
Megachile (Xanthosarus) latimanus Say, 1823	1	0.02	6	Megachilidae	N	P	C	S	L	P
Osmia (Melanosmia) bucephala Cresson, 1864	4	0.08	24	Megachilidae	N	P	C/P	S	L	P
Osmia (Melanosmia) distincta Cresson, 1864	1	0.02	6	Megachilidae	N	P	C/P	S	M	P
Osmia (Melanosmia) pumila Cresson, 1864	20	0.38	65	Megachilidae	N	P	C/P	S	M	B
Osmia (Melanosmia) atriventris Cresson, 1864	3	0.06	12	Megachilidae	N	P	C/P	S	M	P
Apis (Apis) mellifera Linnaeus, 1758	256	4.80	100	Apidae	E	P	C^1	E	L	B
Bombus (Pyrobombus) bimaculatus Cresson, 1863	8	0.15	41	Apidae	N	P	C^1	E	L	B
Bombus (Pyrobombus) impatiens Cresson, 1863	234	4.39	100	Apidae	N	P	C^1	E	L	B
Bombus (Pyrobombus) perplexus Cresson, 1863	4	0.08	24	Apidae	N	P	C^1	E	L	B
Bombus (Pyrobombus) vagans Smith, 1854	4	0.08	18	Apidae	N	P	C^1	E	L	P
Bombus (Separatobombus) griseocollis (DeGeer, 1773)	10	0.19	41	Apidae	N	P	C^1	E	L	B
Ceratina (Zadontomerus) calcarata Robertson, 1900	207	3.88	82	Apidae	N	P	P	B	S	B
Ceratina (Zadontomerus) dupla Say, 1837	18	0.34	41	Apidae	N	P	P	B	S	B
Ceratina (Zadontomerus) mikmaqi (Rehan + Sheffield, 2011)	6	0.11	29	Apidae	N	P	P	B	S	P
Ceratina (Zadontomerus) strenua Smith, 1879	495	9.29	100	Apidae	N	P	P	B	S	B
Ceratina (Zadontomerus) undet.	8	0.15	6	Apidae	N	P	P	B	S	P
Melissodes (Eumelissodes) agilis Cresson, 1878	3	0.06	18	Apidae	N	O	S	S	M	P
Melissodes (Eumelissodes) subillatus LaBerge, 1961	1	0.02	6	Apidae	N	P	S	S	M	P
Melissodes (Eumelissodes) trinodis Robertson, 1901	2	0.04	6	Apidae	N	P	S	S	M	P
Melissodes (Heliomelissodes) desponsus Smith, 1854	1	0.02	6	Apidae	N	O	S	S	M	P
Melissodes (Melissodes) bimaculatus (Lepeletier de Saint Fargeau, 1825)	3	0.06	18	Apidae	N	P	S	S	L	P
Nomada articulata Smith, 1854	7	0.13	24	Apidae	N	P	[S]	P	S	P
Nomada australis Mitchell, 1962	1	0.02	6	Apidae	N	P	[S]	P	S	P
Nomada bidentate	1	0.02	6	Apidae	N	P	[S]	P	M	P
Nomada cressonii Robertson, 1893	1	0.02	6	Apidae	N	P	[S]	P	S	P
Nomada illinoensis Robertson, 1900	1	0.02	6	Apidae	N	P	[S]	P	S	P
Nomada luteoloides Robertson, 1895	1	0.02	6	Apidae	N	P	[S]	P	M	P
Nomada maculata Cresson, 1863	8	0.15	35	Apidae	N	P	[S]	P	M	P
Panurginus potentillae (Crawford 1916)	2	0.04	12	Apidae	N	P	S	S	S	P
***Peponapis (Peponapis) pruinosa* (Say, 1837)**	9	0.17	29	Apidae	N	O	S	S	L	P
Xylocopa (Xylocopoides) virginica (Linnaeus, 1771)	16	0.30	53	Apidae	N	P	W	B	L	B

[a] Abundance is the total number of specimens collected for both years across all sites.

[b] % of Individuals is the percentage of total collection identified as this species ($n = 5{,}331$).

[c] % Sites is the percentage of sampled yards with this species ($n = 17$ sites).

[d] Origin: Each species is classified as native (N) or exotic (E) to North America based on Cane (2003).

[e] Pollen specificity: Each species is classified as either oligolectic (O; a pollen specialist collecting pollen from a single plant family or genus) or polylectic (P; a pollen generalist collecting pollen from multiple plant families).

[f] Nest substrate: Classification of the nest substrate of each species. Soil (S), cavity (C), soft/rotting wood (SW), wood (W), or pith (P); nest substrates in brackets indicate the host of a parasitic species. Nesting preference annotated with C1 for cavities in preexisting, constructed or manmade burrows or crevices to house reproductive chambers (e.g. Cane et al. 2007).

[g] Behavior: Classification of the nesting behavior of each species. Solitary or communal (S), subsocial (B), eusocial (E), or parasitic (P).

[h] Size is the intertegular distance classification. < 1.5 mm = Small (S), 1.6–3 mm = Medium (M), > 3.1 mm = Large (L). When only males collected, no measurements taken (–).

[i] Method is the bee capture method. Hand-netted (H), pan traps (P), both hand-netted and pan traps (B).

Underlined ecological characteristics indicate when species characteristics were inferred from closely related species.

Species in bold represent county records.

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We captured five species in the genus Sphecodes, which are primarily parasites of others in its family *Halictidae* (Sheffield et al. 2003).

Lawn Flora

We recorded 63 different flowering plant species in 17 lawns during 2013 and 2014. Dandelion (*Taraxacum officinale*) was the most widespread flower, found in all lawns in both years (Table 4). White clover (*Trifolium repens*), purple violet (*Viola sororia*), yellow wood-sorrel (*Oxalis stricta*), Canadian horseweed (*Conyza canadensis*), annual fleabane (*Erigeron annuus*), dwarf cinquefoil (*Potentilla canadensis*), and Pennsylvania smartweed (*Polygonum pensylvanicum*) were recorded in at least 60% of all sites for the two years (Table 4). In 2013, horseweed, hairy rock cress (*Arabis hirsute*), and white clover represented more than 67% of all flowers, whereas in 2014, white clover, yellow wood-sorrel, purple smartweed and purple violet were the most abundant species (Table 4). A third of the flower species recorded were native to North America whilst 60% of the flowers have origins in Europe, Asia, and or Africa. The remaining 6% of the plants either had a worldwide distribution or were from South America. In 2013, the majority of flowers were from North America; however, in 2014, there wasn't a clear majority (Table 4).

In the quadrat vegetation surveys, we recorded an additional 38 nonflowering species growing within the Springfield lawns. Kentucky bluegrass (*Poa pratensis*), *Zoysia* grasses, smooth crabgrass (*Digitaria ischaemum*), and fine fescues (*Festuca* spp.) were the most widespread and dominant species growing in the lawns, and with the exception of crabgrass, most likely intentionally planted. Bare soil was present in 16 of the 17 yards. Although the mean percent of the plots classified as bare soil was relatively low (6.3% in 2014 and 8% in 2013), some of the yards were estimated as having up to 27.9% classified as bare soil.

Table 3. Ecological characteristics summarized for bees collected in Springfield, MA, in 2013 and 2014

Taxonomic / ecological grouping	No. of species	Total indiv	% Species ($n = 111$)	% Indiv ($n = 5{,}331$)
Family				
Colletidae	6	42	5.4	0.8
Halictidae	49	3,751	44.1	70.4
Andrenidae	19	157	17.1	2.9
Megachilidae	12	74	10.8	1.4
Apidae	25	1,307	22.5	24.5
Exotic / Native				
Exotic	6	316	5.4	5.9
Native	105	5,015	94.6	94.1
Floral specificity				
Oligolectic	11	147	9.9	2.8
Polylectic	100	5,184	90.1	97.2
Nest substrate				
Cavity / Pith	14	82	12.6	1.5
Cavity[1]	6	516	5.4	9.7
Wood	1	16	0.9	0.3
Pith	5	744	4.5	14.0
Soil	81	3,940	73.0	73.9
Soft / rotting wood	3	20	2.7	0.4
undet	1	13	0.9	0.2
Behavior				
Solitary or communal	54	819	48.6	15.4
Parasitic	14	38	12.6	0.7
Eusocial	38	3,712	34.2	69.8
Subsocial	5	750	4.5	14.1
Body size				
Small (<1.5 mm)	50	3,821	45.0	71.9
Medium (1.6–3 mm)	48	938	43.2	17.6
Large (>3.1 mm)	13	558	11.7	10.5

Discussion

We encountered an abundant and diverse bee fauna in lawn-dominated yards in Springfield, MA, with >5,300 bee specimens, and 111 species representing 29% of the bee species reported for Massachusetts (J. Milam, personal observation). The general expectation for urban biodiversity, including bees, is a homogenized community dominated by exotic species (McKinney 2006). However, homogenization might be scale dependent (i.e., city scale; Groffman et al. 2014), and intensive field studies have the potential to dispel our notion of cities as depauperate of rich animal communities. This diversity of bees suggests that bees live within the vicinity of the lawns and that lawns may provide floral and nesting resources, as confirmed by our hand-netting surveys which targeted lawn flowers, and that the majority of our bees were small-bodied.

The species diversity and degree of specialization of the bee communities at our study sites was comparable to other studies of bees in suburban yards. Although Fetridge and colleagues (2008) recorded similar species richness (110 spp.), they focused sampling on the flowerbeds, which were set within a matrix of large, well-manicured lawns that varied in use of herbicides, pesticides, and commercial fertilizers. Fetridge et al. (2008) did not sample for bees in the lawns because lawn flowers were sparse due to frequent mowing and in some yards, there was the added factor of the application of herbicides. Frankie et al. (2005) recorded 76 bee species in their survey of flower-rich gardens that included a mix of native and exotic plants in two northern California suburbs. A large proportion of the California bees captured by Frankie et al. (2005) were generalists, primarily explained by the abundance of exotic plants. A Vancouver, British Columbia, study

Table 4. Lawn flower diversity and geographical origin for yard in Springfield, MA, in 2013 and 2014

Origin	No. of species	% Species ($n = 63$)	Total abundance	% Flower abundance	% Flower abundance 2013 ($n = 52{,}770$)	% Flower abundance 2014 ($n = 113{,}637$)	Total abundance 2013	Total abundance 2014
Asia	2	3.2	3,658	2.2	0.1	0.5	42	535
Eurasia	18	28.6	15,262	9.2	7.1	10.2	3,720	11,542
Eurasia / Africa	5	7.9	51,314	30.8	18.3	35.9	9,721	41,593
Europe	13	20.6	4,714	2.8	2.5	3.0	1,323	3,391
North America	21	33.3	54,853	33.0	63.9	18.6	33,718	21,135
South America	1	1.6	7,642	4.6	0.6	6.5	317	7,325
Worldwide	3	4.8	23,249	14.0	5.8	17.8	3,081	20,168

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surveying four different urban land-uses, including flower gardens in backyards, recorded 56 bee species (Tommasi et al. 2004). In Chicago, Tonietto et al. (2011) compared green roofs with prairies and parks and recorded 63 species, though the green roofs had the lowest richness and abundance. Urban bee communities such as those from the studies listed above and from Springfield tend to be generalist bees that are able to forage on exotic flowering plants. Nonetheless, these and other urban bee studies demonstrate the importance of urban green spaces in general and residential yards in particular in supplementing the urban bee communities (Hinners et al. 2012, Baldock et al. 2015, Threfall et al. 2015).

Studies of bee communities in urban and suburban areas differ with respect to the abundance of exotic bee species in these habitats. Our results were similar to suburban yards in New York (Fetridge et al. 2008) as well as northern California yards (Frankie et al. 2005), yet contrasted with the findings of Matteson et al. (2008), who reported a much higher percentage of exotic bees in New York City community gardens. The New York City community gardens differed with regards to the urban matrix, whereas these small garden plots were located amidst more intensely developed areas compared to suburban yards, and thus could explain the differences in exotic species (Matteson et al. 2008). We collected few Hylaeus spp., none of which were exotic. In contrast to Matteson et al. (2008) whose most abundant species collected were exotic Hylaeus (*H. leptocephalus*, *H. hyalinatus*), and the less common *H. punctatus* (representing 20% of all individuals). Fetridge et al. (2008) recorded the presence of *H. hyalinatus,* which represented 4% of individuals in NY suburban gardens. The European honeybee was the most abundant and widespread exotic species collected in our study. Although we did not detect any backyard beekeeping within proximity of any of our sites, honeybees can travel up to 6 km during the course of foraging activities (Beekman and Ratnieks 2000). It is also possible honeybees at our sites could be wild-nesting naturalized bees.

Suitable nesting substrates can act as a limiting resource to bee diversity and abundance for some populations of bees (Potts and Willmer 1997, Potts et al. 2005). Many ground-nesting bees prefer loose, well-drained sandy or loamy soils (Cane 1991) in exposed sunny spots. Our study location is on the extensive, relatively flat-topped outwash plains and deltaic sand and gravel sediments deposited into glacial Lake Hitchcock by the Chicopee River (F. R. Morrison, personal communication). Soils derived from these glacial deposits provide suitable substrate for ground-nesting bees. Indeed, almost 75% of all bees collected in this study nested in soil, including the sand-specialists *L. pilosum*, *Agapostemon texanus*, and *L. tegulare*. The number of ground-nesting bees in our study was much higher than the two New York studies (Fetridge et al. 2008, Matteson et al. 2008). The majority of bees in community gardens consisted of cavity nesting species, largely represented by exotic *Hylaeus* species (Matteson et al. 2008). It is suggested that the higher reported abundance of cavity-nesting species in urban settings and remnant habitat fragments compared to natural or suburban habitats, could be the presence of available manmade nesting sites (Hernandez et al. 2009). However, for our study only 18% of all bee spp. and 11.2% of all individual bees were cavity or pith-nesters (not including bees using preexisting cavities or manmade burrows). The low number of cavity nesting bees in our study is interesting, but might be linked to a tornado that destroyed many buildings, trees, and vegetation in 2011 that were subsequently removed as part of the cleanup effort.

Although not abundant in the Springfield yards, the availability of dead or dying vegetation such as blackberries and raspberries (*Rubus* sp.) and sumac (*Rhus* sp.) encourage pith-nesting species like the highly abundant *Ceratina strenua*. Pith-nesting *Ceratina* species were well represented in our study, particularly *C. strenua*, which represented 9.3% of all bees captured and found in all of the 17 yards. In contrast, *C. strenua* was not reported from Matteson et al. (2008) and represented only 2.7% of the bees in the study by Fetridge et al. (2008). Although we did not evaluate the abundance of *Rubus* and *Rhus* spp. in surrounding yards, we suspect that they are present nearby. The presence of cavity and pithy-stem nesting bee species in our samples suggests that, although the lawns do not provide required nesting substrates, they do provide foraging resources. We suggest future studies to target nesting behaviors of suburban bees to better understand how nesting resources might be limiting or whether bees take advantage of additional novel nesting substrates.

Because bees depend upon flowers for food and nest provisioning, flowering plants are a dominant mechanism for structuring bee communities. It follows that the distribution of specialist or oligolectic bees are restricted by the distribution of their hosts (Potts et al. 2003). Bee communities in our study were dominated by polylectic species. This pattern aligns with patterns found in other urban bee studies (e.g., Fetridge et al. 2008, Matteson et al. 2008, Frankie et al. 2005, Tonietto et al. 2011) that document bees foraging on flowers planted to enhance yard aesthetics. Nevertheless, we collected 11 oligolectic species. Of note was the abundant and widespread *Lasioglossum oenotherae* ($n = 125$, recorded in 88% of all yards), a bee specializing on evening primrose (Onagraceae), followed by *Peponapis pruinosas* ($n = 9$), a specialist on squash flowers (*Cucurbita* L.). Less abundant specialist bees captured included two early spring bees *Andrena erythrogaster* ($n = 3$), and *A. frigida* ($n = 1$) specializing on willows (*Salix* L.); and *A. carolina*, a specialist on blueberries (*Vaccinium* L.), and *Colletes latitarsis*, a specialist on ground cherry (*Physalis* L.). The low numbers of Colletidae spp. captured in our study is likely a factor of our not collecting on or the lack of appropriate flowers visited by Colletidae with floral preferences. The Asteraceae specialists included *A. asteris*, *A. helianthi*, *A. placata*, *Melissodes desponsa*, *M. subillata*, and *M. trinoidis*. Host plants for these bees were not found in the Springfield yards, but these bee species represented medium-bodied sized bees and therefore have foraging ranges that might encompass neighboring yards (Greenleaf et al. 2007).

Foraging distances are found to vary with environmental conditions such as floral density and distribution (Ricketts 2001). The majority of the bees in our study were polylectic, small-bodied bees with small foraging ranges, likely on the same order of magnitude as the size of the study lawns (Greenleaf et al. 2007) and thus were conceivably supported by the abundant floral resources within the study yards, and reinforced by the hand-netted bees. The medium- to large-bodied bees with larger foraging distance potential could have flown into the yards from nearby yards or other urban green spaces searching for floral resources. We suggest the need for additional research on the nutritional value of the pollen from the lawn flowers to further our understanding of this potentially valuable resource.

The most abundant bee in our study, *L. illinoense* had not been recorded in Massachusetts since roughly the 1920s, based on a single record from the Museum of Comparative Zoology at Harvard University, collected in Boston, MA, by C. W. Johnson (M. Veit, personal communication), presumably between 1903 and 1932 when he was the Principal Curator for the Boston Society of Natural History (Gray 1933). This southern species was abundant in New York and Chicago surveys (Fetridge et al. 2008, Matteson et al.

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2008, Tonietto et al. 2011), but not found in New York preserves (Fetridge et al. 2008).

Conservation Implications

Suburban lawns in our study provided resources to a surprisingly high number of bees, building on prior research that urban green spaces have an important role to play for urban bee conservation (Baldock et al. 2015). Our study provides baseline information on the resources provided by lawn-dominated yards embedded within a suburban matrix. Fortel and colleagues (2014) conducted a two-year monthly sampling effort along an urban gradient in France and recorded a third of all the wild bees from the country, with a peak in diversity at more intermediate levels of development. As part of the Natural History Museum of Los Angeles County's Biodiversity Science: City and Nature (BioSCAN) investigation, scientists discovered 30 unknown species of *Megaselia* flies in Los Angeles backyards (Hartop et al. 2015). Together, these studies suggest that there is habitat potential within urban green spaces.

The floral richness we recorded implies that untreated lawns are not as resource-poor as previously thought. The spontaneous plants might provide pollen and nectar resource for suburban bees (Potts et al. 2003, Larson et al. 2014). Our study suggests that when not intensively managed, these flowers can serve as wildlife habitat and contribute to networks of urban green spaces (Threlfall et al. 2015). Tommasi and colleagues (2004) found that dandelions and other weedy species in unmaintained areas of Vancouver had the most bee species visiting these plants. By investigating the pollen contents in trap nests for two cavity-nesting bees in Toronto, MacIvor et al. (2014) found that pollen from clover (*Trifolium repens*) dominated the nests and consisted of as much as 65% of the pollen collected. In addition to clover's capacity to fix nitrogen, it also has a relatively high protein value (35.4%; Roulston et al. 2000). None of the lawns in our study were treated with chemical herbicides, which by their nature eliminate flowering plants from lawns and would likely reduce their suitability for bees. Our results are consistent with other studies that suggest that less intensively managed lawns may provide suitable habitat for ground-nesting bees (Threlfall et al. 2015), by allowing for the presence of bare patches of soil and reduced compacted soils that can discourage ground nesting bees (Tonietto et al. 2011). Thus, developing outreach to homeowners and lawn care companies to encourage rather than eliminate lawn flowers such as dandelions and clover and thin grass cover or bare spots could be a key strategy for urban bee conservation programs targeting private yards (Larson et al. 2014).

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