To mow or to mow less: Lawn mowing frequency affects bee abundance and diversity in suburban yards

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ABSTRACT

Green spaces embedded within the urban matrix, particularly residential yards, could mitigate negative aspects of urban development and provide pollinator habitat. Lawns represent a dominant green space, and their management consists of frequent mowing to inhibit the growth of ostensibly "weedy" species (e.g., dandelions and clover). Since widespread population declines of bees and other pollinators from habitat loss are a growing concern, these spontaneous flowers could provide pollen and nectar sources throughout the growing season. We experimentally tested whether different lawn mowing frequencies (1, 2 or 3 weeks) influenced bee abundance and diversity in 16 suburban western Massachusetts yards by increasing lawn floral resources. Lawns mowed every three weeks had as much as 2.5 times more lawn flowers than the other frequencies. Interestingly, lawns mowed every two weeks supported the highest bee abundance yet the lowest bee richness and evenness. We suggest these patterns were driven by a combination of more abundant floral resources (compared with 1-week yards), easier access to lawn flowers due to shorter grass and a more drastic impact on grass biomass and floral resources (compared with 3-week yards), and the dominance of a few generalist bees overwhelming our samples, thus driving richness and evenness. Our results highlight a "lazy lawnmower" approach to providing bee habitat. Mowing less frequently is practical, economical, and a timesaving alternative to lawn replacement or even planting pollinator gardens. Given the pervasiveness of lawns coupled with habitat loss, our findings provide immediate solutions for individual households to contribute to urban conservation.

1. Introduction

Bees and other pollinators provide essential ecosystem services in agricultural and pristine landscapes (Gallai et al., 2009; Ollerton et al., 2011), and are experiencing severe declines on a global scale (Vanbergen et al., 2013). Loss and alteration of habitat primarily due to urban development together with the intensification of agricultural practices (e.g., increased applications of pesticides, tilling, monocultures, reduced season-long floral resources) largely contribute to these declines (Goulson, 2013; Harrison and Winfree, 2015; Vanbergen et al., 2013; Winfree et al., 2009). However, recent urban research has documented cities supporting a surprising level of bee richness and abundance (e.g., Fischer et al., 2016; Frankie et al., 2005; Harrison and Winfree, 2015; Matteson et al., 2008; Pardee and Philpott, 2014; Threlfall et al., 2015), suggesting that public parks, ruderal grasslands, meadows, community gardens and flower gardens in private yards have the capacity to serve as

bee refugia (Hall et al., 2017). Some cities may even harbor more diverse and abundant populations of native bees compared with nearby forest preserves and other natural systems (Baldock et al., 2015; Fetridge et al., 2008; Winfree et al., 2007). Consequently, green spaces embedded within the urban matrix could mitigate negative aspects of urban development, by providing pollinator and other wildlife habitat (Goddard et al., 2010). However, it is unclear how bees respond to one of the most pervasive urban green spaces, lawns.

Lawns cover $> 163,000 \, km^2$ in the US and include golf courses, athletic fields, commercial and industrial parks and urban and suburban yards (Milesi et al., 2005). High proportions of lawns are located in yards, and serve both social and environmental functions. From a historical and social perspective, the lawn represented a status symbol of upward mobility and more recently, a platform for self-expression of, or projecting adherence to social norms (Nassauer et al., 2009; Robbins, 2007; Robbins and Sharp, 2003).

Fig. 1. Example of a lawn-dominated yard participating in the study. Note the minimal landscaping and bare patches in the lawns, which were common throughout the sites. Also note the yard sign in the lawn explaining the objectives of the study. Not only did this demonstrate a 'cue to care' (Nassauer et al., 2009) but also informed neighbors about their role in improving the sustainability of their neighborhoods.

Lawns also provide important ecosystem services. Depending on soil texture, storm water can infiltrate pervious lawns and can serve as a reservoir for some of the run-off (Mueller and Thompson, 2009). Lawns might also mitigate the urban heat island by regulating humidity, particularly when irrigated (Hall et al., 2016). A suburban lawn's capacity for storing carbon (C) and nitrogen (N) can exceed that of non-urban grasslands (Pouyat et al., 2006; Raciti et al., 2008).

In addition to ecological benefits, the intensive management that lawns require can negatively impact urban and suburban ecosystems. Typical lawn management consists of irrigating, applying chemicals and mowing, and is carried out by millions of individual households and neighborhood associations whose actions have ecological and social consequences (Cook et al., 2012). For example, Americans use up to 48 gal of water per day for irrigating lawns and gardens (Environmental Protection Agency; www.epa.gov/waterwise). In arid regions, this kind of water use diminishes scarce natural resources. In addition, fertilizers and other chemical applications can degrade water quality and contaminate groundwater (Law et al., 2004), while gas-powered lawn mower exhaust fumes elevate CO_2 emissions (Zirkle et al., 2011). Although not every household irrigates or fertilizes (Polsky et al., 2014), most households mow to conform to societal expectations, city ordinances, and the personal satisfaction of a neat and tidy yard (Robbins, 2007). Many municipalities even enforce 'weed laws' to ensure conformity of the lawn ideal by restricting grass height (e.g., a Chicago ordinance prohibits lawn vegetation from exceeding 24.4 cm; Municipal Code of Chicago: §7–28-120). Intensive lawn management requires time and financial commitments, and are often driven by aesthetics and social norms to adhere to ideals of orderly, weed-free, lush carpets of green grass (Jenkins, 1994; Nassauer, 1995; Nassauer et al., 2009; Robbins, 2007).

One of the outcomes of frequent lawn mowing is a simplistic vegetation configuration. Consequently, many ecologists and wildlife organizations have dismissed the habitat potential of lawns, referring to these lawn-dominated yards as 'sterile environments for biodiversity' (Gaston et al., 2005: 3342). However, even with it's simplicity, lawns can support rich and diverse plant communities. A survey of 52 residential lawns in Sheffield, UK recorded 159 species of vascular plants

(Thompson et al., 2004). However, floral richness and abundance in these lawns might depend on lawn management behaviors and disturbance (Bertoncini et al., 2012; Grime, 1974; Wastian et al., 2016). Research on bees in New York residential yards that had extensive flower gardens showed that frequent lawn mowing (and herbicide application) depleted lawns of floral resources for bees (Fetridge et al., 2008), suggesting that less frequent mowing and avoiding herbicides could have the opposite effect.

Lawns lacking applications of herbicides and other chemicals generally support spontaneous flowers, such as common dandelion *Taraxacum officinale* (Asteraceae) and white clover *Trifolium repens* (Fabaceae) (Bertoncini et al., 2012). This has potential habitat implications for bees (Larson et al., 2014) given their dependence on pollen and nectar resources from flowering plants (Frankie et al., 2005). However, frequent (e.g., weekly) lawn mowing generally prohibits plants from flowering (Fetridge et al., 2008). Because declines in native bees and other pollinators are largely caused by habitat loss (Vanbergen et al., 2013), nectar and pollen from these and other 'weedy' species have the potential to support bee conservation in urban areas. In this study, we manipulated lawn mowing behaviors in suburban yards to test the hypothesis that decreasing mowing frequency may result in increased lawn floral resources, and in turn, increased bee abundance, bee richness and bee diversity. Testing the effects of alternative lawn care management practices on floral resources may have important implications for bee and other pollinator populations given the cumulative area of lawns in urban and suburban areas in the U.S. and the millions of people that manage these systems.

2. Material and methods

2.1. Study sites

We conducted the study in 16 single-family, owner-occupied suburban yards (sites) in Springfield, Massachusetts, USA. Because we were working with private households, we relied on volunteers that we recruited via a local tree planting organization. Parcels ranged in size between 0.03 and 0.18 ha (typical of medium-density housing stock within Springfield), and houses were built between 1921 and 1957. We

required that the yards not be treated with herbicides or irrigated during the study, or have designated pollinator or vegetable gardens. The sites were predominantly comprised of lawns with limited cultivated floral resources or hedges (Fig. 1). All sites were at least 500 m apart with the exception of two sites, which were across the street from each other. We assessed the degree of spatial autocorrelation among the yards with Mantel tests using bee abundance, which confirmed their independence (r = 0.14, $p = 0.13$).

2.2. Mowing regimes

Lawns were mowed from May through September in 2013 and 2014, using a Toro 19″ self-mulching push mower, (mowing height set at 6.35 cm). Grass clippings remained on the lawn. We assigned each yard to a mowing frequency regime: mowed every seven days (one-week; n = 8 yards), 12–14 days (two-weeks; n = 7 yards), or 18–21 days (three-weeks; n = 8 yards) to represent the range of typical mowing behaviors (one to two weeks) to a more extreme frequency (three weeks; Robbins, 2007). Seven yards participated in both years of the study and thus these repeat yards were assigned a different mowing regime for the second year of the study. To ensure households adhered to the experimental restrictions (e.g., frequency and height of mowing), we provided a free lawn mowing service and mowed all participating lawns for the duration of the study.

2.3. Floral resource and vegetation estimates

We limited our floral measurements to dicots, rather than monocots. Floral resource abundance for lawn and yard flowers was estimated prior to each bee-sampling event (n = five per year; see Section 2.4). We counted the total number of flowers per species in bloom within the entire property boundary. We classified flowers as either 'yard flowers' (i.e., planted ornamental cultivars or hybrids such as azaleas to account for other floral resources not impacted by mowing) or 'lawn flowers' (i.e., flowers growing spontaneously amidst the planted turf grass such as dandelions to test the impacts of mowing). For flower identification we used Del Tredici (2010), Peterson and McKenny (1996) and the online GoBotany resource (https://gobotany.newenglandwild.org). For flowers we were unable to identify, we labeled them 'unknown1', 'unknown2', etc., to ensure total abundance was calculated per sampling round. We then took the sum of all yard flowers and all lawn flowers to calculate a total yard and lawn floral abundance per site per sampling event. Composites were treated as one flower. Prior to each mowing event we also measured the grass height in three random locations and then calculated a mean grass height for each site and each sampling round. In a Geographical Information System (GIS), we calculated canopy cover within 50 m of each site and total lawn area for each parcel (i.e., yard). For the canopy cover, we used Google Earth imagery from 2014. We centered our calculation at the intersection of the driveway and road to serve as a proxy for landscape-level vegetation effects for each yard. This aerial coverage also coincides with the published travel distance of the smallest bees we collected (Greenleaf et al., 2007; Appendix A1). To calculate total lawn area, we used a Google Earth image from 2012 with 'leaf-off' aerial view and parcel data from the City of Springfield. For both images, the scale was 1:1250, and minimum mapping unit was 3 m. To calculate the percent of bare ground in the lawns, we conducted two intensive sampling events per site, per year using the quadrat sampling method. The plots consisted of three 1 m² plots per site whereby we assigned a percent coverage of bare soil for the plot, and then calculated a mean bare ground percentage for the parcel.

2.4. Bee sampling

We collected bees approximately every three weeks in 2013 and 2014, with five sampling periods per site per year. Regardless of

treatment (every one, two or three-week regime), bees were collected on warm, sunny days, prior to mowing using a combination of pan traps and hand-held insect nets. Pan traps consisted of 30 plastic 3.25 oz. (96 ml) cups (Solo brand, model number p325w) painted florescent yellow, blue or left white, and were filled with soapy water (the detergent reduced the surface tension). For each sampling round in each yard, and following the methods outlined in Droege (2008), we strategically placed 30 pan traps (10 arrays of three pan traps, one of each color), near lawn flowers in areas of the yard that received full sun for the majority of the day. Rather than random transects, this method allowed us to optimize trap deployment within these small yards, which were often fragmented by driveways, fences, houses and other structures. Pan trap arrays were placed in roughly the same location within each site during the course of the season. Pan traps were left out for 24 h. Because pan traps bias the collection of smaller bees (Cane et al., 2000), we also swept for bees using hand nets for 15 min. The sweepnet survey occurred immediately following the collection of the pan traps and prior to mowing. The contents of the 30 pan traps and each sweep event were transferred into separate plastic whirl-pak bags, containing 70% ethanol alcohol. In the lab the bees were washed, dried, pinned, and labeled. Bees were identified to the species level when possible using a number of different keys (Gibbs, 2010, 2011; Gibbs et al., 2012; Mitchell, 1960, 1962), identified by J. Milam, S. Droege, and M. Veit, and databased. Specimen vouchers are deposited with the US Forest Service Urban Natural Resources Institute.

2.5. Data analysis

2.5.1. Bee community composition

We conducted all analyses in R 3.2.3 (R Core Team, 2014) unless otherwise noted. For the bee community composition analyses, we aggregated the bee data at the site-level and calculated total richness per site and species composition per site. We used sample-based rarefaction curves and 95% confidence intervals to assess differences in bee species richness among the three lawn mowing treatments using the *specaccum* function in the *vegan* package of R (Oksanen et al., 2016). We estimated the slope of the rarefaction curve at seven sites, which was the minimum number of sites sampled per treatment over both years, since the accumulation curves did not reach asymptotes. We used nonmetric multidimensional scaling (NMDS) with the *MASS* package in R (Venables and Ripley, 2002) to visualize bee community dissimilarities among the three mowing treatments. By maximizing the rank correlations between the calculated dissimilarity distance matrix and the plotted distances, NMDS, a method for visualizing the data in multi-dimensional space, reduces samples in ordination space while preserving ecological differences (McCune and Grace, 2002). We used the Bray-Curtis distance measurement and conducted a Monte Carlo randomization test on the final stress value to calculate a goodness of fit. We then conducted a one-way analysis of similarity (ANOSIM) to test the null hypothesis of no differences in the bee community among the treatments using the *anosim* function in the *vegan* package of R (Oksanen et al., 2016). ANOSIM converts distances into ranks to calculate the differences between inter and intragroup rank dissimilarities (Clarke, 1993; Clarke and Green, 1988). We only included bees recorded at more than one site. We repeated this analysis for different functional guilds (bee origin, nesting substrate, behavior, body size; Lerman and Milam, 2016) and for the ten most abundant bees.

2.5.2. Effects of lawn mowing frequency on vegetation and bees in suburban yards

We used a generalized linear mixed effects modeling (GLMM) framework to assess differences in mean grass height, lawn flower abundance, yard flower abundance, bee abundance, and bee evenness among the three lawn mowing treatments (n = 5 separate models). To calculate abundance, we summed the total number of flowers and bees per site per sampling round per year (round-level: five sampling events

per site per year). Bee evenness was calculated for each site per sampling round per year using the Buzas-Gibson's evenness e^H/S statistic in Past 3.11 (Hammer et al., 2001). The yard flower, lawn flower and bee abundance mixed models were fitted with a negative binomial distribution because of the overdispersed structure of the data using the *glmmADMD* package in R (Fournier et al., 2012). The bee evenness and mean grass height mixed models were fitted using the *gls* and *lme* functions in the *nlme* package (Pinheiro et al., 2016). We included a yard flower abundance model to account for other available floral resources, and to ensure that yard floral resources did not differ among treatments.

Model selection for random effects included testing for significance of random intercepts, random intercepts and/or random slopes, autocorrelation structures, and variance structures using the protocol recommended by Zuur et al. (2009) and described by Contosta et al. (2011). Potential random intercepts included year, sampling round (n = five rounds per year to account for seasonal differences), date (to account for weather effects) and site ID (to account for parcel size, bare soil and other site-specific differences). Potential random slopes included treatment. We fit all possible combinations of random intercept and random slope terms, including multiple, nested random intercepts with or without random slopes. For all models that reached convergence, the optimal random effects structure (random intercept and/ or slope) was determined with Akaike's Information Criteria for small sample sizes (AICc; Burnham et al., 2011). For *lme* models, potential autocorrelation structures included first-order autoregressive and autoregressive moving average structures while potential variance structures consisted of unequal variance across treatment groups. The fit of candidate autocorrelation and variance structures was also determined with AICc. If neither variance covariates nor autocorrelation structures improved model fit, then the data were considered to have met the assumptions of equal variance and equal correlation necessary for linear modeling (Littell et al., 1998). Using this protocol, we found that none of the model fits improved with the addition of variance or autocorrelation structures, but several models were enhanced with random effects. Random effect structures evaluated for each modeled response variable are listed in Appendix A2.

Fixed effects for each model included mowing frequency, with some models also containing hypothesized mowing treatment covariates. The lawn flower model included a covariate for mean grass height, both alone and as an interaction term with treatment. Similarly, the bee abundance and bee evenness models included covariates for grass height, lawn flower richness, and lawn flower abundance, both singly and interacting with treatment. Selection of fixed effects occurred with a backward selection procedure. First, we determined the significance of two-way interactions between lawn mowing treatment and treatment covariates. This consisted of fitting models with and without the two-way interaction, where models that contained multiple two-way interactions (bee abundance and bee evenness) each had the two-way interaction removed in turn. Model fit with and without interaction terms was determined using log-likelihood ratio tests (lawn flower abundance and bee abundance models using GLMM) or maximum likelihood (bee evenness and mean grass height models using *lme*), together with AICc (Zuur et al., 2009). If *p* was > 0.05, then the two-way interaction was considered not significant and was removed from the model. Selection of single fixed effects (treatment and covariates) was similar, in that each effect was omitted in turn and evaluated for its contribution to overall model fit.

Due to relatively small sample sizes, we used a bootstrapping procedure to obtain both the model-level *p*-values of differences across treatments as well as pairwise *p*-values of differences between treatments. Bootstrapping was based on the method described by Warton and Hui (2011), in which we resampled the data with replacement from the null distribution (i.e., we resampled across the entire data set, not within treatment groups), ran the full and null model (that did not contain treatment) with the resampled data, performed the likelihood

ratio test between the full and reduced model, and repeated for 1000 iterations. Significance of the full model (differences among all treatments) was assessed as the ratio of *p*-values from the bootstrapped null distribution that were less than the *p*-value obtained from the likelihood ratio test of the original data. The procedure for determining pairwise differences between treatment groups was similar, in that the full model was run using the bootstrapped null distribution 1000 times, each using the one, two or three-week treatments as the model base cases in turn. This was necessary because the model base case automatically defaults to the one-week treatment. Thus, ensuring the two-week treatment and three-week treatment were the model base cases in turn allowed for comparison of all treatments to one another. Significant differences from the model base case were evaluated with the ratio of *p*-values obtained with the *summary* function from the null distribution to the *p*-values determined with the *summary* function from the original data.

2.5.3. Relationships between vegetation and bee variables and site characteristics

To understand whether factors independent of mowing might impact floral resources, bee abundance, and bee evenness, we evaluated the relationships between our response variables and site characteristics (lawn size and percent bare ground) and the landscape habitat composition adjacent to the study sites (percent canopy cover). Because these features did not vary over the course of the study, we performed this analysis with data aggregated within each site and year (site-level data). Thus we fitted five 'site characteristic effects' models in which grass height, yard flower abundance, lawn flower abundance, bee abundance, and bee evenness were response variables, and lawn size and canopy cover were predictor variables. Selection of random and fixed effects was the same as for the models evaluating the effects of mowing frequency. Because all final models for this portion of the analysis were null models that contained no significant effects, we did not perform any bootstrapping to obtain *p*-values. Instead, *p*-values reported are determined by fitting models with and without each fixed effect in turn and comparing them with tests of log-likelihood (GLMM models) and maximum likelihood (lme models). Scripts developed to apply these statistical models to the data can be downloaded via the GitHub digital repository (https://github.com/Contosta/lawn-bees).

3. Results

3.1. Bee community composition

The effect of mowing on the bee assemblage was evident in total bee abundance and bee richness. We collected a total of 4587 bees representing 93 species during the ten sampling rounds (see Appendix A1 for complete list and associated life history traits including origin, nesting substrate, behavior and body size). Summarized bee abundance and richness in relation to lawn mowing treatments were as follows: weekly mowing = 1425 bees representing 72 species, mowed every two weeks = 1903 bees representing 60 species, and mowed every three weeks = 1259 bees representing 62 species (Fig. 2, Appendix A1). Our rarefaction analysis reflected these overall abundance and richness values, showing lower bee richness in the two-week treatment. The slopes (*m*) for each treatment did not reach asymptotes after sampling all seven sites per treatment (*m* = 2.34, 1.89, and 2.03 for 1, 2, and 3 weeks respectively; Fig. 2), indicating that additional species may be detected with additional sampling, particularly in one-week treatments (highest slope). There were no discernable bee community patterns among the three mowing treatments for the entire bee community (NMDS observed stress = 0.128, ANOSIM; R = −0.006, *p* = 0.505; Fig. 3), bee origin (NMDS observed stress = 0.018, ANOSIM; R = 0.023, *p* = 0.281), nesting guild (NMDS observed stress = 0.039, ANOSIM; R = 0.023, *p* = 0.308), behavior (NMDS observed stress = 0.016, ANOSIM; R = 0.022, *p* = 0.303), bee size (NMDS observed stress = 0.015, ANOSIM; R = 0.018, *p* = 0.291), or for the ten
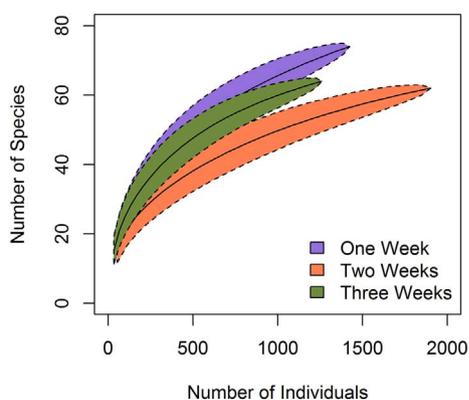
Fig. 2. Sample-based rarefaction curves of bee species richness as a function of number of individuals for three mowing frequencies (one-week, two-week and three-week treatments). Solid lines represent the estimated number of species for each treatment, and the shaded area between the dashed lines indicates the estimated 95% confidence interval.
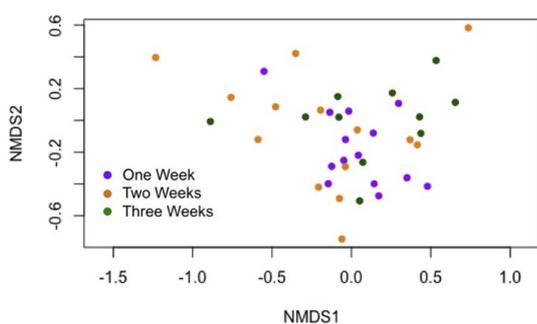

Fig. 3. Non-metric multidimensional scaling (NMDS) of the bee community in suburban yards using a Bray-Curtis similarity index. Only species with at least two observations were included. There were no discernable differences in the bee community among mowing treatments.

most abundant bees (NMDS observed stress = 0.087, ANOSIM; R = 0.037, p = 0.185). For the lawn flowers, we identified 37 species in lawns mowed weekly (plus 3 unidentified species), 28 species in lawns mowed every two weeks (plus 7 unidentified species) and 34 species for lawns mowed every three weeks (plus 6 unknowns; Appendix A3). These totals do not include species unidentified.

3.2. Effects of lawn mowing frequency on vegetation and floral resources

Lawns mowed every three weeks had significantly taller grass (prior to mowing), compared with the one and two-week treatments (p = 0.002, Fig. 4a, Tables 1, 2, Appendix A4). Likewise, lawn floral abundance varied with mowing frequency (p = 0.035), with lawns mowed every three weeks having significantly more lawn flowers compared with the weekly and every two-week treatment (Fig. 4b, Tables 1, 2). As for yard floral abundance, there was no significant difference among all three mowing treatment frequencies (p = 0.207, Tables 1, 2). All models included treatment as a fixed effect. Final model selections for vegetation, floral resources, bee abundance, bee evenness and site-level factors are listed in Appendix A2.

3.3. Effects of lawn mowing frequency on bee abundance and bee evenness

Bee abundance differed among mowing treatments (p = 0.002), such that lawns mowed every two weeks had significantly more bees compared with the weekly and every three-week treatment (Fig. 4c, Tables 1, 2). Bee evenness also varied with mowing frequency (p < 0.0001); lawns mowed every two weeks had significantly lower evenness values than the other two treatments (Fig. 4d, Tables 1, 2).

3.4. Relationships between vegetation and bee variables and site characteristics

Neither floral resources nor measures of the bee community (evenness and abundance) varied as a function of site characteristic effects (e.g., bare ground, canopy cover and lawn size; p > 0.05, Table 2).

4. Discussion

By manipulating lawn mowing frequency, we established that lawns with the three-week mowing treatment had significantly greater floral abundance than the one or two-week treatments, and that the two-week mowing regime supported the highest bee abundance yet the lowest bee richness and evenness (Fig. 2; Fig. 4b, c, d). With the inclusion of site-level (i.e., yard flowers in planted beds and lawn size) and neighborhood influences (i.e., percent canopy cover) in our models, we accounted for some of the inherent variation found in yards and lawns. The lack of influence from these variables further suggested that the differences we observed in lawn flower and bee abundance and diversity resulted from mowing frequency and not underlying drivers of biodiversity such as vegetation cover. Although lawns cannot provide the same level of habitat quality, function, and continuity of natural areas (Wastian et al., 2016), they do have floral resources for sustenance and pollen for provisioning nests, therefore their management (i.e., lawn mowing) may represent a dominant mechanism for structuring urban bee communities (Frankie et al., 2005; Potts et al., 2003).

We recognize that results from a small sample in a single ecoregion might not be applicable to other lawn-dominated systems. However, our bootstrap analysis provides additional confidence in the interpretation of our findings for a New England suburb. Studies from other ecoregions with even smaller sample sizes found management effects on biodiversity. For example, research from southeastern Pennsylvania, USA compared six yards landscaped with native plants against six yards with conventional landscaping. The authors found that the native yards attracted higher abundances of native lepidopteran larvae and subsequently, insectivorous birds, thus increasing native bird diversity (Burghardt et al., 2009). We suggest future studies simultaneously investigate multiple metropolitan areas embedded within different climates, ecosystems and different landscaping practices to better assess the generality of our findings (Groffman et al., 2017), and to further advance our understanding of how lawn management behaviors might influence pollinator diversity at broader scales.

4.1. Diverse and abundant suburban bee communities

For this study, we documented 93 species of bees collected from the lawn-dominated yards (Appendix A1). These 93 species represent roughly a quarter of bee species recorded in Massachusetts, include 14 Massachusetts county records, and featured the highly abundant *Lasioglossum illinoense*, a species not recorded in Massachusetts since 1920 (Lerman and Milam, 2016). Other urban bee studies have also amassed impressive species lists (e.g., Baldock et al., 2015; Fetridge et al., 2008; Matteson et al., 2008; Pardee and Philpott, 2014; Tommasi et al., 2004), dispelling the notion that cities are "biological deserts" and support findings that bees can be abundant and diverse in urban settings (Hall et al., 2017). In addition to being primarily native species and soil-nesters, the majority of the Springfield bees were small-bodied (Appendix A1), suggesting that these short-distance fliers took advantage of the floral resources in the study lawns, especially yards mowed every two weeks (Greenleaf et al., 2007).

Although the suburban lawns we sampled featured a highly diverse bee community, the ten most abundant species represented 78% of all captures. Of this top ten, the exotic *Apis mellifera* only represented 5%. Likewise, the large-bodied *Bombus impatiens* was widespread throughout our study but only accounted for a small percentage of the
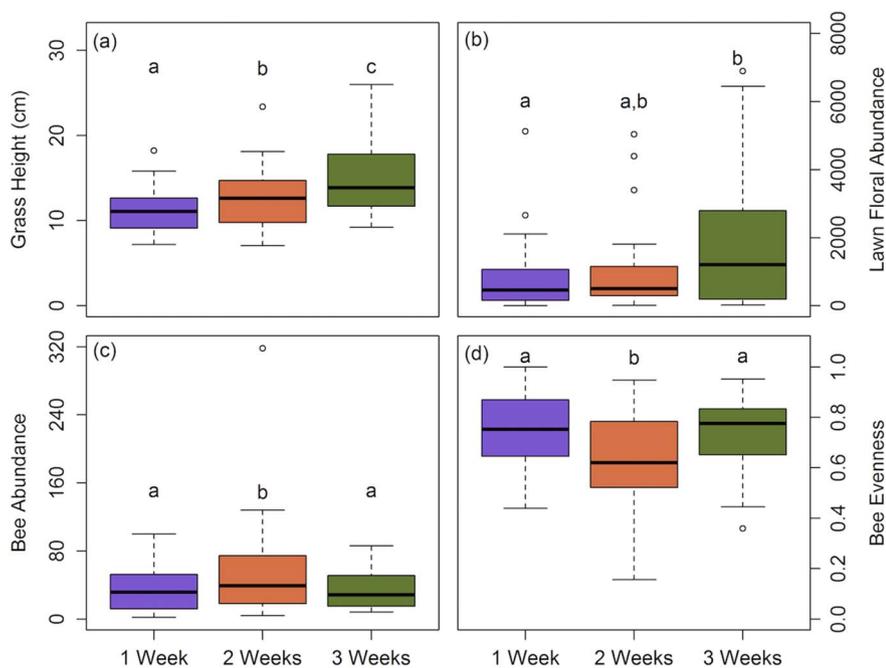
Fig. 4. Box and whisker plots of the relationships between lawn mowing frequency and mean grass height (a), lawn floral abundance (b), bee abundance (c), and bee evenness (d). Yard floral abundance was not significant and not included. The top and bottom of each box indicate values at the 25th and 75th percentile, the bold line indicates the median, and whiskers extending beyond the box depict data within 1.5 times the interquartile range with individual circles indicating potential outliers in the distribution tail. Bee data include individuals captured from the pan traps and sweep net surveys. Lowercase letters above boxes indicate pairwise differences between treatment groups.

bees. Collectively, however, the small-bodied sweat bees (i.e., *Lasioglossum* spp.) dominated our samples, accounting for 42% of all bees collected. In addition to the abundant *L. illinoense*, the soil nesting *L. pilosum* and *L. tegulare*, and the small-bodied but pith-nesting *Ceratina strenua* were present in large numbers, suggesting ample nesting opportunities present in the study yards or surrounding yards. We recommend future studies further explore the different nesting opportunities and how they vary among yards and management regimes. This could further our understanding of some of the limiting factors structuring urban bee communities.

4.2. Effects of lawn mowing frequency on bee evenness and abundance

Mowing frequency altered the evenness of bees within suburban yards, though the patterns we observed did not fully support our hypothesis, in that lawns mowed every week and every three-weeks had higher evenness (Fig. 4d) and richness (Rarefaction curves; Fig. 2) when compared with the two-week treatment. However, much of the species richness documented was due to singleton captures (Appendix A1). The dominance of a few species together with many singletons was expected (Fisher et al., 1943). However, the dominance of a few species coupled with high bee abundance might have overwhelmed our sampling efforts and provided fewer chances for encountering rare or uncommon species in the two-week lawns (Gotelli and Colwell, 2001). We further suggest that perhaps because of the high species richness in combination with the dominance of a few species, the NMDS failed to discern differences in community assemblages among the three treatments. The lower diversity we encountered in two-week sites may not translate to reduced ecosystem function. Recent research suggests that only a few, common bee species, regardless of species richness, provide the bulk of pollination services in agricultural ecosystems (Winfree et al., 2015). The same might hold true for suburban lawns such that bee abundance, particularly of generalist species, outweighs richness in supporting ecosystem services in urban settings. Bees and other animals pollinate 87% of all flowering plants (Ollerton et al., 2011). Thus their persistence and abundance in cities and suburbs is paramount for ecosystem function, quality of life, and human health (Threlfall et al., 2015) since pollinator-dependent trees, shrubs, garden plants, and spontaneous flowers present in urban and suburban green spaces provide shade, reduce air pollution, increase property value, and enhance wildlife

habitat (Akbari et al., 2001; Lerman et al., 2014). We suggest that future studies address the relative importance of species-abundance distribution in urban systems to better understand the role of pollination for supporting ecosystem services (Williams and Winfree, 2013).

Mowing frequency also altered the abundance of bees within suburban yards, though not exactly as predicted. Although the three-week treatment had the highest lawn floral abundance (Fig. 4b), the two-week treatments had the highest bee abundance (Fig. 4c), partially supporting our hypothesis that lawn management behaviors increase lawn floral resources, and in turn, bee abundance. One possible explanation for the differences in bee abundance between the one and two-week treatments is via the resource-matching rule (RMR; Parker, 1978; Pulliam and Caraco, 1984) in that bee abundance matched lawn flower abundance (Frankie et al., 2005). Although not significant, the lawns mowed every week had fewer lawn flowers than the lawns mowed every two weeks (Appendix A4) and hence, might explain why there were fewer bee captures. We did not anticipate the lower bee abundance for the lawns mowed every three-weeks. According to the RMR, bees in the three-week yards might have under-matched the abundant lawn flowers and underexploited this resource-rich habitat. Our results concur with a modeling exercise demonstrating that individual foragers consistently under-matched resource availability (Kennedy and Gray, 1993). Alternatively, bees in the two-week treatment might be over-matching resources (Shochat, 2004). The lower abundance of bees captured in the three-week treatment might also suggest that the bees had plenty of alternative resources (i.e., abundant lawn flowers), making the pan traps less attractive and hence lower captures (Lerman et al., 2012a; Morris and Davidson, 2000). We did not analyze the nectar quality of the lawn flowers, which might provide additional insight to our unexpected results. However, other studies have documented a mismatch in floral density and floral quality in that the same plant species growing at different densities or in isolation might lack the same resource quality or visitation rates (Shapiro, 1975; Tepedino and Stanton, 1981). We identified 54 species growing spontaneously in the study lawns with white clover by far the most abundant and wide-spread, and *Conzya* spp. (horseweed), *Polygonum pensylvanicum* (Pennsylvania smartweed) and *Oxalis dillenii* (yellow wood sorrel) also plentiful (Appendix A3). However, lawn floral richness was not selected in any of our models, suggesting that it had little influence on our results. Regardless, little information exists on the nutritional

Table 1

Pairwise comparisons testing the effects of different lawn mowing treatments on mean grass height, lawn and yard flower abundance, bee abundance, and bee evenness. Significant differences from the model base case were evaluated with the ratio of p-values obtained with the *summary* function from the bootstrapped null distribution to the p-values determined with the *summary* function from the original data. Because the model base case automatically defaults to the one-week treatment, we performed this test three times, releveling the model base case for the two-week and then the three-week treatment to compare all treatments to one another. Asterisks indicate significance at α = 0.05.

	Response	p-Value
Mean grass height (cm)	Base case (treatment = 1 week)	
	(Intercept)	0.548
	Treatment 2 weeks	0.019*
	Treatment 3 weeks	<0.0001***
	Relevel (treatment = 2 weeks)	
	(Intercept)	0.928
	Treatment 1 week	0.018*
	Treatment 3 weeks	0.027*
	Relevel (treatment = 3 weeks)	
	(Intercept)	0.961
	Treatment 2 weeks	0.027*
	Treatment 1 week	0.026*
Lawn flower abundance	Base case (treatment = 1 week)	
	(Intercept)	1.000
	Treatment 2 weeks	0.413
	Treatment 3 weeks	0.019*
	Relevel (treatment = 2 weeks)	
	(Intercept)	1.000
	Treatment 1 week	0.384
	Treatment 3 weeks	0.126
	Relevel (treatment = 3 weeks)	
	(Intercept)	1.000
	Treatment 2 weeks	0.126
	Treatment 1 week	0.018*
Yard flower abundance	Base case (treatment = 1 week)	
	(Intercept)	0.993
	Treatment 2 weeks	0.356
	Treatment 3 weeks	0.582
	Relevel (treatment = 2 weeks)	
	(Intercept)	0.987
	Treatment 1 week	0.340
	Treatment 3 weeks	0.092
	Relevel (treatment = 3 weeks)	
	(Intercept)	0.941
	Treatment 1 week	0.607
	Treatment 2 weeks	0.091
Bee abundance	Base case (treatment = 1 week)	
	(Intercept)	0.998
	Treatment 2 weeks	<0.0001***
	Treatment 3 weeks	0.347
	Relevel (treatment = 2 weeks)	
	(Intercept)	0.996
	Treatment 1 week	0.003**
	Treatment 3 weeks	0.010*
	Relevel (treatment = 3 weeks)	
	(Intercept)	0.999
	Treatment 1 week	0.344
	Treatment 2 weeks	0.012*
Bee evenness	Base case (treatment = 1 week)	
	(Intercept)	0.998
	Treatment 2 weeks	0.001**
	Treatment 3 weeks	0.617
	Relevel (treatment = 2 weeks)	
	(Intercept)	1.000
	Treatment 1 week	0.001**
	Treatment 3 weeks	0.002**
	Relevel (treatment = 3 weeks)	
	(Intercept)	1.000
	Treatment 1 week	0.562
	Treatment 2 weeks	0.002**

Table 2

Effects of lawn mowing frequency (treatment) and site characteristics (lawn size, bare ground, and percent canopy cover) on mean grass height, lawn and yard flower abundance, bee abundance, and bee evenness. Significance was determined as the proportion of p-values obtained from the bootstrapped null distribution that were less than the p-value obtained from the likelihood ratio test of the original data. Asterisks indicate significance at α = 0.05.

	Response	p-Value
Mowing frequency effects	Mean grass height	
	Treatment	0.002**
	Lawn flower abundance	
	Treatment	0.035*
	Yard flower abundance	
	Treatment	0.207
	Bee abundance	
	Treatment	0.002**
	Bee evenness	
	Treatment	<0.0001***
Site characteristic effects	Mean grass height	
	Lawn size	0.093
	Bare ground	0.251
	Canopy cover	0.945
	Lawn flower abundance	
	Lawn size	0.204
	Bare ground	0.282
	Canopy cover	0.089
	Yard flower abundance	
	Lawn size	0.503
	Bare ground	0.244
	Canopy cover	0.217
	Bee abundance	
	Lawn size	0.227
	Bare ground	0.369
	Canopy cover	0.49
	Bee evenness	
	Lawn size	0.987
	Bare ground	0.392
	Canopy cover	0.773

The differences in grass height among the treatments may also explain why bee abundance was highest in sites mowed every two weeks. The taller grass in the three-week treatments (mean height: 15.1 cm compared with 11.2 cm and 12.5 cm for one and two-weeks respectively; Fig. 4a, Appendix A4) may have prohibited access to the flowers, rendering the floral-abundant lawns less attractive. For example, *Lasioglossum illinoense* is responsible for much of the increase in abundance for the two-week treatments (Appendix A1). Perhaps a combination of abundant flowers coupled with the medium length grass height in the two-week lawns provided easier access to the floral resources compared to the conditions in the three-week lawns.

Alternatively, the lawn flowers might lack performance traits necessary for competing with the tall grass, particularly in the three-week treatments, leading to pollen limitation and hence less attractive habitats for bees (Knight et al., 2005). A three-week mowing hiatus had the most drastic effect on floral abundance and grass height, and the decrease in bee abundance in the three-week treatment might be reflective of this level of disturbance (Armesto and Pickett, 1985). After being mown, all lawns regardless of frequency had the same grass height since the mower height was set at 6.35 cm for all mowing events. However, grasses in the three-week treatment had on average 8.75 cm removed after each mowing event compared with 4.85 and 6.15 for one and two-week treatments respectively (Appendix A4) and three-week lawns went from on average 1900 lawn flowers to very few. Although we did not count the number of flowers remaining after mowing, very few flowers persisted (S. Lerman, personal observation). Thus, lack of access to floral resources in yards, reduced performance of lawn flowers, and/or the intensity of the disturbance from the three-week mowing regime may also help to explain why bee abundance was lower in the three week yards than might have been expected based on lawn flower abundance alone.

value of these and other spontaneous flowers. We suggest future studies conduct detailed observations of visitation rates on spontaneous plants to further our understanding of whether and how bees might take advantage of the many floral resources available.

We selected three mowing frequencies for our study to mimic existing lawn management behaviors (every week to two weeks) and a more extreme, but a plausible mowing frequency (three-week treatment). We did not include even more intense lawn mowing frequencies such as two or three times per week or less intense management regimes such as once per season (in essence, a no-mow treatment), nor did we alter mowing height. Further, the assigned mowing conditions were only for one growing season. These alternative conditions on a longer time scale might further differentiate the patterns we documented in addition to capturing time lags. For example, Tepedino and Stanton (1981) suggest that floral abundance more closely aligned with abundance of nest cells provisioned and not necessarily adult bee abundance. Although half of our sites were sampled for the two years of the study, we altered mowing frequency between the years to better control for site effects and explicitly test for different lawn mowing frequencies. A better understanding of carryover effects from previous years could further elucidate the relationship between floral resources, nest success and adult bee abundance (Potts et al., 2003). Nonetheless, mowing every two weeks supported >30% more bees (Appendix A1) than the other mowing frequencies.

4.3. Bee conservation in suburban yards

We targeted yards with few cultivated floral resources to better tease apart the habitat potential of the lawn flowers on bees. Although all participating sites had some yard flowers (generally cultivars and hybrids), we failed to document strong effects on the bee community (Table 1). Nonetheless, programs like the National Pollinator Garden Network's Million Pollinator Garden Challenge program (www.millionpollinatorgardens.org) holds tremendous promise, and we recognize that cultivated flowers and pollinator gardens could be important in yards, particularly those with more abundant plantings (Baldock et al., 2015). However, our results align with other recent urban bee studies and suggest that yards lacking specialized plantings and pollinator gardens but have spontaneous lawn flowers also support bees (Garbuzov et al., 2015; Fischer et al., 2016). For example, Larson et al. (2014) documented 25 species of pollinators visiting white clover and 21 species visiting common dandelion in Kentucky, USA lawns. Further, in a nesting study from Toronto, Canada, clover was the dominant pollen source for successful brood cells for two generalist bee species (MacIvor et al., 2014), suggesting that these plants have high protein content (Roulston et al., 2000). As with these and other investigations, our study suggests that in addition to pollinator gardens, the unintentional and spontaneous flowers growing in lawns without chemical inputs also support bees, and serve as unsung heroes for bee conservation.

Several conservation organizations promote the removal and replacement of lawns with native plants (e.g., Cornell Lab of Ornithology's Habitat Network www.yardmap.org) to support native wildlife. In situations where replacing lawns with native plants is not practical, our study suggests that altering mowing behaviors by reducing lawn mowing frequency to a two-week regime may improve bee habitat and increase bee abundance in urban areas. We recognize some of the limitations of extrapolating results from our study. Results are from a single northeastern US city and only encompassed two growing seasons. Nonetheless, our results support other studies that documented strong responses from wildlife to alternative management behaviors. For example, creating 'uncut refuges' in sections of managed hay meadows in Switzerland provided a continuous supply of pollen and nectar for native bees (Buri et al., 2014). Similarly, delayed mowing in the Champlain Valley of New York and Vermont USA significantly increased breeding success for grassland birds nesting in hayfields, primarily by reducing mortality due to mowing during the vulnerable nestling stage (Perlut et al., 2006). The management implications from different mowing regimes from both private yards and agricultural systems can be applied to other lawn-dominated landscapes, for example golf course margins, public parks and highway verges. If coordinated, then even a small percentage of adoptees of lower mowing frequency, delayed mowing, or no mowing can scale up and might have positive conservation implications for bee habitat (Goddard et al., 2010; Hall et al., 2017).

4.4. The lazy lawnmower

We suggest a 'lazy lawnmower' approach as an additional option for managing yards for wildlife. The recommendation to mow lawns less frequently to help promote bee conservation might garner broad public support (potentially compared with lawn reduction or replacement) because it more closely aligns with current single-family homeowner motivations for adopting lawn-dominated yardscapes. A New England study on lawn care attitudes and behaviors found that householders were concerned about water quality and thus were willing to try alternative lawn care approaches that were more environmentally friendly (e.g., higher mowing height, reduced fertilizer application). However, the authors also noted significant barriers to changing behaviors including concerns that the alternative lawn care would not be as aesthetically pleasing, would incur additional financial burdens, and would require more time for upkeep (Eisenhauer et al., 2016). Likewise, in a multi-city survey, respondents ranked various landscaping decisions in which aesthetically pleasing, weed-free, and ease of maintenance topped the list, while provisioning for wildlife ranked fifth out of eight choices (Larson et al., 2015). These studies suggest that wildlife-friendly landscaping has some support, but the acceptance of weeds and the inclusion of more native plants (which are not as showy as their non-native congeners; Frankie et al., 2005) might be at odds with more preferred management goals of aesthetics and ease of maintenance (Lerman et al., 2012b; van Heezik et al., 2012). Based on our interactions with participating households and their neighbors, our treatment of a three-week mowing frequency appeared unkempt and exceeded the tolerance of many homeowners and their neighbors, and thus the two-week regime might reconcile homeowner ideals with pollinator habitat.

Moving towards a mechanistic approach when studying urban biodiversity (Shochat et al., 2006) increases our ability to directly link management with ecological outcomes, and ultimately lead to effective action. Our experimental approach demonstrated how altering lawn management decisions influences bee abundance despite the inherent variability present in suburban yards. Manipulating lawn mowing behaviors also demonstrates a new and creative approach for supporting urban biodiversity by rethinking the role lawns play towards enriching urban areas. Mowing less frequently is practical, economical, environmental and a timesaving alternative to lawn replacement or even planting pollinator gardens, that has the potential to be widely adopted if it can overcome barriers to social acceptance. Most importantly, our research shows that individual households can contribute to urban conservation.

Acknowledgements

Appendix A

Appendix A1

Comprehensive list of bees collected during the ten sampling rounds (five per year), associated abundances according to lawn mowing frequency, and natural history traits. Origin: each species is classified as native (N) or exotic (E) to North America based on Cane (2003). Pollen: pollen specificity classification as either oligolectic (O; a pollen specialist colleting pollen from a single plant family or genus) or polylectic (P: a pollen generalist collecting pollen from multiple plant families); [P] indicates parasitic species that do not collect pollen. Nest: 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 C[1] for cavities in preexisting, constructed or manmade burrows or crevices to house reproductive chambers. Behavior: classification of the nesting behavior of each species. Solitary or communal (S), subsocial (B), eusocial (E) or parasitic (P). Bold indicates the ten most abundant species.

Species	1 wk	2 wks	3 wks	Family	Origin	Pollen	Nest	Behavior	Size
Colletes latitarsis Robertson, 1891	1	0	0	Colletidae	N	O	S	S	M
Hylaeus (Hylaeus) mesillae (Cockerell, 1896)	6	0	1	Colletidae	N	P	C	S	S
Hylaeus (Prosopis) affinis/modestus	5	0	4	Colletidae	N	P	C	S	S
Agapostemon (Agapostemon) sericeus (Förster, 1771)	6	0	3	Halictidae	N	P	S	S	M
Agapostemon (Agapostemon) texanus Cresson, 1872	5	16	18	Halictidae	N	P	S	S	M
Agapostemon (Agapostemon) virescens (Fabricius, 1775)	74	29	54	Halictidae	N	P	S	S	M
Augochlora (Augochlora) pura (Say, 1837)	0	1	0	Halictidae	N	P	SW	S	M
Augochlorella aurata (Smith, 1853)	5	3	5	Halictidae	N	P	S	E	S
Halictus (Nealictus) parallelus Say, 1837	1	1	2	Halictidae	N	P	S	E	M
Halictus (Odontalictus) ligatus Say, 1837	111	79	57	Halictidae	N	P	S	E	M
Halictus (Protohalictus) rubicundus (Christ, 1791)	9	7	2	Halictidae	N	P	S	E	M
Halictus (Seladonia) confusus Smith, 1853	102	111	109	Halictidae	N	P	S	E	S
Lasioglossum (Dialictus) anomalum (Robertson, 1892)	1	0	0	Halictidae	N	P	S	E	S
Lasioglossum (Dialictus) bruneri (Crawford, 1902)	5	17	6	Halictidae	N	P	S	E	M
Lasioglossum (Dialictus) coreopsis (Robertson, 1902)	2	0	0	Halictidae	N	P	S	E	S
Lasioglossum (Dialictus) cressonii (Robertson, 1890)	6	3	5	Halictidae	N	P	SW	E	S
Lasioglossum (Dialictus) ellisiae (Sandhouse, 1924)	2	1	3	Halictidae	N	P	S	E	S
Lasioglossum (Dialictus) ephialtum Gibbs, 2010	57	75	32	Halictidae	N	P	S	E	S
Lasioglossum (Dialictus) fattigi (Mitchell, 1960)	1	0	0	Halictidae	N	P	S	E	S
Lasioglossum (Dialictus) illinoense (Robertson, 1892)	112	594	195	Halictidae	N	P	S	E	S
Lasioglossum (Dialictus) imitatum (Smith, 1853)	7	26	22	Halictidae	N	P	S	E	S
Lasioglossum (Dialictus) izawsum Gibbs, 2011	0	1	0	Halictidae	N	[P]	[S]	P	S
Lasioglossum (Dialictus) laevissimum (Smith, 1853)	0	0	2	Halictidae	N	P	S	E	S
Lasioglossum (Dialictus) leucocomum (Lovell, 1908)	24	7	12	Halictidae	N	P	S	P	S
Lasioglossum (Dialictus) lineatulum (Crawford, 1906)	3	1	0	Halictidae	N	P	S	E	S
Lasioglossum (Dialictus) oceanicum (Cockerell 1916)	2	0	0	Halictidae	N	P	S	E	S
Lasioglossum (Dialictus) oblongum (Lovell, 1905)	0	1	0	Halictidae	N	P	SW	E	S
Lasioglossum (Dialictus) pilosum (Smith, 1853)	210	316	172	Halictidae	N	P	S	E	S
Lasioglossum (Dialictus) planatum Gibbs, 2011	1	0	3	Halictidae	N	P	S	E	S
Lasioglossum (Dialictus) platyparium (Robertson 1895)	0	0	1	Halictidae	N	[P]	[S]	P	S
Lasioglossum (Dialictus) smilacinae (Robertson, 1899)	2	2	3	Halictidae	N	P	S	E	S
Lasioglossum (Dialictus) subvirdatum (Cockerell, 1938)	0	1	0	Halictidae	N	P	S	E	S
Lasioglossum (Dialictus) taylorae Gibbs, 2010	1	0	0	Halictidae	N	P	S	E	S
Lasioglossum (Dialictus) tegulare (Robertson, 1890)	86	69	74	Halictidae	N	P	S	E	S
Lasioglossum (Dialictus) versans (Lovell, 1905)	1	0	0	Halictidae	N	P	S	E	S
Lasioglossum (Dialictus) versatum (Robertson)	0	0	1	Halictidae	N	P	S	E	S
Lasioglossum (Dialictus) vierecki (Crawford, 1904)	6	8	11	Halictidae	N	P	S	E	S
Lasioglossum (Dialictus) viridatum (Lovell, 1905)	0	1	0	Halictidae	N	P	S	E	S
Lasioglossum (Dialictus) weemsi (Mitchell, 1960)	12	13	9	Halictidae	N	P	S	E	S
Lasioglossum (Evylaeus) cinctipes (Provancher, 1888)	0	1	1	Halictidae	N	P	S	E	M
Lasioglossum (Hemihalictus) foxii (Robertson, 1895)	0	1	0	Halictidae	N	P	S	S	S
Lasioglossum (Hemihalictus) nelumbonis (Robertson, 1890)	1	2	0	Halictidae	N	P	S	S	S
Lasioglossum (Hemihalictus) pectorale (Smith, 1853)	26	25	32	Halictidae	N	P	S	S	S
Lasioglossum (Lasioglossum) acuminatum McGinley, 1986	2	0	1	Halictidae	N	P	S	S	M
Lasioglossum (Lasioglossum) coriaceum (Smith, 1853)	4	4	2	Halictidae	N	P	S	S	M
Lasioglossum (Lasioglossum) leucozonium (Schrank, 1781)	14	6	11	Halictidae	E	P	S	S	M
Lasioglossum (Sphecodogastra) oenotherae (Stevens, 1920)	23	55	40	Halictidae	N	O	S	S	M
Lasioglossum undet.	5	5	3	Halictidae	N				
Sphecodes fattigi Mitchell, 1956	1	0	0	Halictidae	N	[P]	[S]	P	S
Sphecodes illinoensis (Robertson, 1903)	2	3	1	Halictidae	N	[P]	[S]	P	S
Sphecodes mandibularis Cresson, 1872	0	2	3	Halictidae	N	[P]	[S]	P	S
Andrena (Andrena) carolina Viereck, 1909	0	0	1	Andrenidae	N	O	S	S	M

Species				Family					
Andrena (Callandrena s.l.) asteris Robertson, 1891	1	0	0	Andrenidae	N	O	S	S	M
Andrena (Callandrena s.l.) helianthi Robertson, 1891	1	0	0	Andrenidae	N	O	S	S	M
Andrena (Callandrena s.l.) placata Mitchell, 1960	0	1	0	Andrenidae	N	O	S	S	M
Andrena (Larandrena) miserabilis Cresson, 1872	0	0	1	Andrenidae	N	P	S	S	M
Andrena (Melandrena) commoda Smith, 1879	1	0	0	Andrenidae	N	P	S	S	M
Andrena (Scrapteropsis) alleghaniensis Viereck, 1907	2	1	0	Andrenidae	N	P	S	S	M
Andrena (Simandrena) nasonii Robertson, 1895	2	0	1	Andrenidae	N	P	S	S	S
Andrena (Taeniandrena) wilkella (Kirby, 1802)	1	1	1	Andrenidae	E	P	S	S	S
Calliopsis (Calliopsis) andreniformis Smith, 1853	25	18	33	Andrenidae	N	P	S	S	S
Anthidium (Anthidium) manicatum (Linnaeus, 1758)	2	1	5	Megachilidae	E	P	C	S	L
Anthidium (Proanthidium) oblongatum (Illiger, 1806)	4	2	3	Megachilidae	E	P	C	S	M
Hoplitis (Alcidamea) producta (Cresson, 1864)	3	1	5	Megachilidae	N	P	P	S	M
Megachile (Eutricharaea) rotundata (Fabricius, 1793)	1	1	3	Megachilidae	E	P	C	S	M
Megachile (Litomegachile) mendica Cresson, 1878	1	3	2	Megachilidae	N	P	C	S	M
Megachile (Megachile) centuncularis (Linnaeus, 1758)	0	2	3	Megachilidae	N	P	C	S	M
Megachile (Megachile) montivaga Cresson, 1878	2	0	0	Megachilidae	N	P	C	S	M
Megachile (Xanthosarus) latimanus Say, 1823	1	0	0	Megachilidae	N	P	C	S	L
Osmia (Melanosmia) atriventris Cresson, 1864	0	0	1	Megachilidae	N	P	C/P	S	M
Osmia (Melanosmia) bucephala Cresson, 1864	0	1	0	Megachilidae	N	P	C/P	S	L
Osmia (Melanosmia) distincta Cresson, 1864	0	0	1	Megachilidae	N	P	C/P	S	M
Osmia (Melanosmia) pumila Cresson, 1864	1	0	4	Megachilidae	N	P	C/P	S	M
Apis (Apis) mellifera Linnaeus, 1758	83	96	65	Apidae	E	P	C^1	E	L
Bombus (Pyrobombus) bimaculatus Cresson, 1863	3	3	2	Apidae	N	P	C^1	E	L
Bombus (Pyrobombus) impatiens Cresson, 1863	82	59	79	Apidae	N	P	C^1	E	L
Bombus (Pyrobombus) perplexus Cresson, 1863	2	1	0	Apidae	N	P	C^1	E	L
Bombus (Pyrobombus) vagans Smith, 1854	1	2	0	Apidae	N	P	C^1	E	L
Bombus (Separatobombus) griseocollis (DeGeer, 1773)	2	4	4	Apidae	N	P	C^1	E	L
Ceratina (Zadontomerus) calcarata Robertson, 1900	59	38	12	Apidae	N	P	P	B	S
Ceratina (Zadontomerus) dupla Say, 1837	9	6	1	Apidae	N	P	P	B	S
Ceratina (Zadontomerus) mikmaqi Rehan and Sheffield, 2011	3	2	1	Apidae	N	P	P	B	S
Ceratina (Zadontomerus) strenua Smith, 1879	167	161	115	Apidae	N	P	P	B	S
Ceratina undet.	5	3	0	Apidae	N	P	P	B	
Melissodes (Eumelissodes) agilis Cresson, 1878	0	2	1	Apidae	N	O	S	S	M
Melissodes (Eumelissodes) subillatus LaBerge, 1961	1	0	0	Apidae	N	P	S	S	M
Melissodes (Eumelissodes) trinodis Robertson, 1901	1	0	1	Apidae	N	P	S	S	M
Melissodes (Heliomelissodes) desponsus Smith, 1854	0	1	0	Apidae	N	O	S	S	M
Melissodes (Melissodes) bimaculatus (Lepeletier de Saint Fargeau, 1825)	1	0	2	Apidae	N	P	S	S	L
Nomada articulata Smith, 1854	1	2	1	Apidae	N	[P]	[S]	P	S
Nomada cressonii Robertson, 1893	1	0	0	Apidae	N	[P]	[S]	P	M
Nomada illinoensis Robertson, 1900	1	0	0	Apidae	N	[P]	[S]	P	M
Panurginus potentillae (Crawford 1916)	0	0	1	Apidae	N	P	S	S	S
Peponapis (Peponapis) pruinosa (Say, 1837)	4	2	3	Apidae	N	O	S	S	L
Xylocopa (Xylocopoides) virginica (Linnaeus, 1771)	7	2	7	Apidae	N	P	W	B	L

Appendix A2

Random effects model selection from the generalized linear mixed model (GLMM and LME) selection process. Summary includes the degrees of freedom (df), Akaike's Information Criteria for small sample sizes (AICc), and the change in AICc (ΔAICc). Random effects include lawn mowing treatment (1, 2, 3 week mowing frequency; treatment), siteID (site), sampling round (round), year and date. Models included no random effects (1), random intercepts for 1, 2, and 3 random effects (e.g., (1|site) + (1|round)), and random slope effects with 1, 2, and 3 random intercepts (e.g., (treatment|site) + (treatment|round) + (treatment|year)).

Response	Model #	Random effects	df	AICc	ΔAICc
Mean grass height	17	treatment|round	10	590.9	0
	11	1|year/date	6	592.2	1
	3	1|round	5	592.6	2
	10	1|round/date	6	593.5	3
	9	1|round/year	6	594.9	4
	14	1|round/year/date	7	595.8	5
	4	1|year	5	597.2	6
	5	1|date	5	599.8	9
	19	treatment|date	10	604.7	14
	18	treatment|year	10	606.8	16

	7	1|site/year	6	610.0	19
	16	treatment|site	10	613.9	23
	1	1	4	617.3	26
	2	1|site	5	617.7	27
	6	1|site/round	6	619.9	29
	8	1|site/date	6	619.9	29
	12	1|site/round/year	7	622.2	31
	13	1|site/round/date	7	622.2	31
	15	1|site/round/year/date	8	624.5	34
Lawn floral abundance	6	(1|site) + (1|round)	9	1775.3	0
	12	(1|site) + (1|round) + (1|year)	10	1776.0	1
	13	(1|site) + (1|round) + (1|date)	10	1776.7	1
	15	(1|site) + (1|round) + (1|year) + (1|date)	11	1777.7	2
	8	(1|site) + (1|date)	9	1782.6	7
	20	(treatment|site) + (treatment|round)	13	1785.2	10
	7	(1|site) + (1|year)	9	1786.4	11
	23	(treatment|site) + (treatment|round) + (treatment|year)	16	1787.4	12
	24	(treatment|site) + (treatment|round) + (treatment|date)	16	1792.0	17
	21	(treatment|site) + (treatment|date)	13	1792.3	17
	2	(1|site)	8@@@	1793.2	18
	16	(treatment|site)	10	1793.6	18
	25	(treatment|site) + (treatment|round) + (treatment|year) + (treatment|date)	19	1795.6	20
	3	(1|round)	8	1796.5	21
	9	(1|round) + (1|year)	9	1798.1	23
	10	(1|round) + (1|date)	9	1798.9	24
	14	(1|round) + (1|year) + (1|date)	10	1800.5	25
	17	(treatment|round)	10	1801.1	26
	22	(treatment|round) + (treatment|year)	13	1804.7	29
	11	(1|year) + (1|date)	9	1806.8	32
	4	(1|year)	8	1808.0	33
	5	(1|date)	8	1808.3	33
	18	(treatment|year)	10	1810.3	35
	19	(treatment|date)	10	1812.2	37
	1	(1)	7	1813.0	38
Yard floral abundance	6	(1|site) + (1|round)	6	1779.5	0
	3	(1|round)	5	1780.1	1
	15	(treatment|date)	7	1780.6	1
	5	(1|date)	5	1780.7	1
	16	(treatment|site) + (treatment|round)	10	1780.7	1
	8	(1|site) + (1|date)	6	1780.8	1
	13	(treatment|round)	7	1781.0	1
	11	(1|site) + (1|round) + (1|year)	7	1781.8	2
	18	(treatment|site) + (treatment|date)	10	1782.0	2
	12	(treatment|site)	7	1782.0	3
	9	(1|round) + (1|year)	6	1782.3	3
	10	(1|year) + (1|date)	6	1782.9	3
	2	(1|site)	5	1783.1	4
	1	(1)	4	1784.8	5
	7	(1|site) + (1|year)	6	1785.4	6
	4	(1|year)	5	1787.0	7
	21	(treatment|site) + (treatment|round) + (treatment|date)	13	1787.7	8
	20	(treatment|year) + (treatment|date)	10	1787.7	8
	19	(treatment|round) + (treatment|year)	10	1788.1	9
	14	(treatment|year)	7	1788.7	9
	17	(treatment|site) + (treatment|year)	10	1789.1	10
	22	(treatment|site) + (treatment|round) + (treatment|year) + (treatment|date)	16	1795.7	16
Bee abundance	12	(1|site) + (1|round) + (1|date)	16	987.0	0
	18	(treatment|site) + (treatment|round)	19	993.4	6
	8	(1|site) + (1|date)	15	995.2	8
	22	(treatment|site) + (treatment|round) + (treatment|date)	22	995.2	8
	6	(1|site) + (1|round)	15	996.6	10
	20	(treatment|site) + (treatment|date)	19	997.0	10
	10	(1|round) + (1|date)	15	997.2	10
	15	(treatment|round)	16	998.7	12
	3	(1|round)	14	999.4	12
	9	(1|round) + (1|year)	15	1001.9	15

	11	(1|year) + (1|date)		15	1005.0 18
	5	(1|date)		14	1005.8 19
	17	(treatment|date)		16	1011.2 24
	21	(treatment|year) + (treatment|date)		19	1016.2 29
	4	(1|year)		14	1052.1 65
	7	(1|site) + (1|year)		15	1054.0 67
	16	(treatment|year)		16	1057.5 70
	19	(treatment|site) + (treatment|year)		19	1061.2 74
	14	(treatment|site)		16	1061.8 75
	1	(1)		13	1062.8 76
	13	(treatment)		13	1062.8 76
	2	(1|site)		14	1064.4 77
Bee evenness	3	1|round		14	−86.4 0
	10	1|round/date		15	−84.7 2
	9	1|round/year		15	−83.7 3
	14	1|round/year/date		16	−82.0 4
	5	1|date		14	−80.6 6
	11	1|year/date		15	−80.1 6
	16	treatment|site		19	−79.0 7
	4	1|year		14	−70.8 16
	18	treatment|date		19	−69.7 17
	1	1		13	−63.5 23
	2	1|site		14	−61.6 25
	6	1|site/round		15	−58.9 27
	7	1|site/year		15	−58.9 27
	8	1|site/date		15	−58.9 27
	17	treatment|year		19	−58.1 28
	12	1|site/round/year		16	−56.2 30
	13	1|site/round/date		16	−56.2 30
	15	1|site/round/year/date		17	−53.4 33
Mean grass height	3	1|year		6	136.9 0
	1	1		5	139.8 3
	4	1|year/site		7	141.1 4
	2	1|site		6	143.5 7
Lawn floral abundance	3	(1|year)		6	455.0 0
	1	(1)		5	455.6 1
	4	(1|year) + (1|site)		7	459.2 4
	2	(1|site)		6	459.3 4
Yard floral abundance	1	(1)		5	460.2 0
	2	(1|site)		6	463.7 4
	3	(1|year)		6	463.9 4
Bee abundance	1	(1)		5	283.5 0
	2	(1|site)		6	286.8 3
	3	(1|year)		6	287.2 4
	4	(1|year) + (1|site)		7	291.0 8
Bee evenness	1	1		5	23.7 0
	2	1|site		6	27.0 3
	3	1|year		6	27.4 4
	4	1|year/site		7	31.6 8

Appendix A3

Comprehensive list of lawn flowers observed during the ten sampling rounds (five per year), and associated abundances according to lawn mowing frequency.

Species	Common name	1 wk	2 wks	3 wks
Achillea millefolium	Yarrow	27	0	50
Allium spp.	Allium	0	0	55
Amaranthus palmeri	Pigweed	100	0	0
Arenaria serpyllifolia	Thyme-leaved sandwort	313	0	0
Barbarea vulgaris	Rocket, yellow	20	0	0
Berteroa incana	Alyssum, hoary	1265	0	0
Centaurea cyanus	Cornflower	0	0	25
Cerastium vulgatum	Mouse-ear chickweed	40	500	554

Chamaecrista nictitans	Wild sensitive plant	35	0	750
Commelina communis	Asiatic dayflower	46	5	491
Conyza spp.	Horseweed	6210	6510	4060
Daucus carota	Queen Anne's lace	0	380	0
Dianthus armeria	Debtford pink	3	0	0
Erigeron annuus	Annual fleabane	403	250	596
Fragaria vesca	Strawberry	5	0	0
Gentiana puberulenta	Downy gentian	0	2	0
Glechoma hederacea	Ground ivy	0	500	0
Glechoma hederacea	Creeping charlie	0	50	400
Hieracium caespitosum	Yellow hawkweed	143	736	156
Houstonia caerulea	Bluet	254	455	0
Hypochaeris radicata	Cats ear	3	0	0
Jacobinia carnea	Kingscrown	0	130	0
Lamium amplexicaule	Henbit	0	40	0
Lepidium sativum	Cress, common white	0	0	50
Linaria vulgaris	Butter-and-eggs	0	0	60
Lotus corniculatus	Birdsfoot trefoil	53	165	655
Lysimachia terrestris	Swamp candle	10	0	0
Mollugo verticillata	Carpetweed	2575	1725	3342
Nuttallanthus canadensis	Blue toadflax	0	0	681
Oxalis stricta	Yellow wood-sorrel	3128	5522	13,935
Plantago lanceolata	Buckhorn plantain	0	25	1950
Plantago spp	Blanchard plantain	0	50	50
Polygonum pensylvanicum	Purple smartweed	5055	3105	2095
Potentila fruticosa	Shrubby cinquefoil	120	0	0
Potentilla argentea	Silvery cinquefoil	175	55	359
Potentilla canadensi	Dwarf cinquefoil	110	800	777
Potentilla simplex	Common cinquefoil	220	0	0
Prunella vulgaris	Heal-all	0	325	0
Rumex acetosella	Red sorrel	0	0	75
Securigera varia	Crown vetch	34	0	197
Silene latifolia	White campion	1	0	0
Stellaria media	Common chickweed	340	25	60
Stellaria spp.	Chickweed sp.	200	100	50
Taraxacum officinale	Dandelion	488	141	739
Thlaspi arvense	Field pennycress	110	660	1845
Tragopogon dubius	Yellow goatsbeard	30	0	15
Trifolium arvense	Rabbit's foot clover	358	0	0
Trifolium campestre	Low hop clover	0	525	5
Trifolium pratense	Red clover	240	196	131
Trifolium repens	White clover	8341	8310	32,936
Veronica spp	Speedwell sp.	20	0	0
Vicia cracca	Cow vetch	150	0	220
Vicia spp	Vetch sp.	0	0	100
Viola sororia	Purple violet	3	0	145
Unknown 1		13	25	4000
Unknown 2		0	0	100
Unknown 3		10	40	25
Unknown 4		200	0	0
Unknown 5		0	0	20
Unknown 6		0	100	0
Unknown 7		0	5	0
Unknown 8		0	530	10
Unknown 9		0	0	200
Unknown 10		0	3	0
Unknown 11		0	5	0

Appendix A4

Summary statistics for response variables (grass height, lawn flower abundance, yard flower abundance, bee richness, bee abundance and bee evenness) and site characteristic variables (lawn area, bare soil percentage and canopy coverage at 50 m) for each lawn mowing frequency (1 week, 2 weeks, 3 weeks) and for the entire study, regardless of treatment.

	Mowing frequency	Grass height (cm)	Lawn flower abundance	Lawn flower richness	Yard flower abundance	Bee richness	Bee abundance	Bee evenness	Lawn area (ha)	Bare soil (%)	Canopy cover (%)
Mean	1 wk	11.2	773.3	5.8	1271.7	11.6	35.6	0.744	0.09	8.2	20.59
	2 wks	12.5	914.2	5.2	685.5	12.0	54.4	0.635	0.06	9.1	24.33
	3 wks	15.1	1893.8	6.2	1986.7	11.0	33.1	0.723	0.07	5.6	18.09
	Study	12.9	1193.8	5.7	1330.6	11.5	40.6	0.703	0.08	7.7	20.86
Minimum	1 wk	6.7	0	1	0	1	2	0.439	0.03	3.9	3.5
	2 wks	7.8	10	1	0	2	4	0.156	0.03	1.0	1
	3 wks	9.4	18	1	0	4	8	0.359	0.03	2.3	0.2
	Study	6.7	0	1	0	1	2	0.156	0.03	0.0	0.2
Maximum	1 wk	18.2	5127	14	6375	20	100	1	0.18	22.5	47.1
	2 wks	23.4	5041	16	5050	20	318	0.947	0.08	27.9	48.9
	3 wks	26.0	6895	10	25,318	23	86	0.951	0.14	13.3	48.9
	Study	26.0	6895	16	25,318	23	318	1	0.18	27.9	48.9
Median	1 wk	11.1	461.5	5.0	732.5	11.5	31.5	0.753	0.08	5.75	12.1
	2 wks	12.4	500.0	5.0	256	13	39	0.621	0.08	6	26.7
	3 wks	14.0	1210.5	6.0	428	10.5	28.5	0.776	0.08	4.5	13.35
	Study	12.3	552.0	5.0	450	11	33	0.73	0.08	5.5	13.7
Standard error	1 wk	0.42	150.24	0.57	234.41	0.81	4.11	0.025	0.02	2.23	6.61
	2 wks	0.59	194.92	0.64	172.18	0.83	9.73	0.033	0.01	3.46	6.20
	3 wks	0.70	333.02	0.36	774.89	0.70	3.22	0.025	0.01	1.44	5.40
	Study	0.36	144.64	0.30	280.57	0.45	3.59	0.016	0.01	1.41	3.40

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