Floral abundance and richness drive beneficial arthropod conservation and biological control on golf courses

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Abstract

Golf courses are among the largest, most ubiquitous highly maintained urban green spaces in the U.S. Although their primary function is recreation, a large portion of golf course land is not actively in use, which presents opportunities to create more ecologically functional areas. Golf courses around the world are converting these out-of-play areas to alternative habitats, but few do so using evidence that maximizes their benefit. Previous studies have investigated effects of conservation plantings on golf courses, but none have isolated the biological control services of flying natural enemies and few investigated effects on native bees. In this study, we ask if converting out-of-play maintained turfgrass areas into wildflower habitats increases bee and insect natural enemy abundance and diversity and their associated ecosystem services. We created wildflower habitats at two levels of species richness and compared their effects on insects within and around them to turfgrass plots. To quantify effects on biological control, we deployed sentinel prey *Spodoptera frugiperda* larvae, excluding ground-dwelling predators. As predicted, we found that converting turfgrasses to flowering habitats increased bees and natural enemies on golf courses. However, high wildflower species richness was an important factor in promoting native bees and natural enemies. We found that wildflower habitats, particularly those with highest floral richness, increased biological control rates in adjacent turfgrass areas by up to 50% compared to turfgrass controls. Our results indicate that purposefully designed conservation plantings on golf courses can help support insect pollinators while also augmenting biological control of turfgrass pests.

Keywords Pollinators · Integrated pest management · Green space · Urban conservation

Introduction

Cities are rapidly expanding and characterized by the replacement and fragmentation of natural vegetation with anthropogenic structures (McKinney 2002; Nowak and Greenfield 2012). Although urbanization can have variable effects on insects (Dale and Frank 2018), most evidence indicates that biodiversity, species richness, and associated ecosystem services decline with increasing urban development (McKinney 2002). Thus, vegetation that remains in cities can provide valuable services like wildlife conservation, air and water filtration, and carbon

sequestration (Dimoudi and Nikolopoulou 2003; Goddard et al. 2010; Nowak et al. 2013). Urban green spaces (e.g., city parks, golf courses, urban gardens) play an important role in conserving wildlife (Colding and Folke 2009; Goddard et al. 2010), benefiting human health (Lee and Maheswaran 2011), and mitigating negative effects of urbanization (Davies et al. 2011; Sun and Chen 2017). Arthropods frequently respond to urban land use change such that higher trophic levels decline (Burkman and Gardiner 2014), pollinator communities change (Hamblin et al. 2017), and some herbivores become more abundant or damaging (Dale and Frank 2018). Since urban green spaces have the potential to provide benefits and are often managed, it is important to develop strategies to increase their ecological value while providing practical pest management solutions for urban land managers (Ahern 2011).

Golf courses are among the largest and most common continuously vegetated managed urban green spaces, averaging 60 ha each in the United States (Lyman et al. 2007). Due to their size and ability to support vigorous plant communities, these spaces can provide valuable arthropod conservation services (Saarikivi et al. 2010; Tanner and Gange 2005). In fact, Mata

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et al. (2017) found that golf courses in Australia supported greater arthropod richness than nearby urban parks or gardens. Despite this, arthropod conservation benefits associated with golf courses are likely under-exploited because courses primarily consist of highly maintained turfgrasses, which lack structural complexity, interspecific diversity, and floral resources. Importantly, between 40 and 70% of U.S. golf course land is not used for the game of golf, yet is largely composed of highly maintained turfgrasses (Lyman et al. 2007). Therefore, utilizing these out-of-play spaces to provide more ecologically functional habitats could increase beneficial wildlife while also reducing maintenance costs (Larson et al. 2017).

There are growing concerns regarding large scale declines in insect biodiversity linked to anthropogenic change (Sánchez-Bayo and Wyckhuys 2019). Insect pollinators in particular are of conservation interest due to their economic and ecological value (Losey and Vaughan 2006) and because many species are sensitive to conditions associated with urbanization (Goulson et al. 2015). Recent evidence indicates that urban heat islands and reduced floral resources associated with urbanization change native bee communities by selecting for those able to use those reduced resources (Biesmeijer et al. 2006; Winfree et al. 2011) or with high thermal tolerance (Hamblin et al. 2017). Fortunately, urban landscapes can support diverse bee assemblages if suitable nesting habitat and resources are available (Fortel et al. 2014; Hall et al. 2017). Golf courses are frequently islands of vegetation among urban development and thus, islands of relatively cooler temperatures and more resources. Although golf courses are often associated with elevated pesticide inputs, which pose risks to beneficial insects (Kunkel et al. 1999; Gels et al. 2002; Larson et al. 2012), following pesticide best management practices and creating conservation habitats in out-of-play spaces poses minimal risk to pollinators (Larson et al. 2017; Dobbs and Potter 2015). In fact, Dobbs and Potter (2015) demonstrated that creating pollinator conservation areas in such spaces attracted pollinators from 51 species within 18 genera, while also reducing golf course maintenance inputs. Therefore, repurposing urban golf course out-of-play areas for pollinator conservation may provide valuable habitat that mitigates the negative effects of urbanization on pollinators.

Importantly, many pollinating insects are also predatory or parasitic flies, wasps, or beetles that attack plant pests. Therefore, providing pollinator conservation habitats can also promote biological control of nearby economically important herbivores (Blaauw and Isaacs 2015). This is particularly important on golf courses where maintenance efforts (i.e., pesticide inputs, fertilizer use) are driven by the need to maintain plant quality and playability, which is often threatened by insect pests (Held and Potter 2012). Several studies have demonstrated that creating conservation habitats translates to increased biological control of insect pests on nearby plants (Blaauw and Isaacs 2015; Dobbs and Potter 2016; Frank and Shrewsbury 2004). Notably,

vegetation diversity (Altieri 1999), structural complexity (Shrewsbury and Raupp 2000), and patch size (Blaauw and Isaacs 2012) each affect arthropod natural enemies and their services. Additionally, others have demonstrated that specific wildflower species support specialist natural enemies that attack key pests of turfgrasses on golf courses (Abraham et al. 2010; Arevalo and Frank 2005; Portman et al. 2010; Rogers and Potter 2004). For example, plantings of *Spermacoce verticillata* L. (Rubiaceae) increased the abundance of *Larra bicolor* (Hymenoptera: Crabronidae), a specialist parasitoid of mole cricket pests, and its associated parasitism rates up to 200 m away (Portman et al. 2010). Therefore, tailoring golf course out-of-play areas to optimize natural enemy conservation may translate to fewer pests, pesticides, and natural resource inputs in these highly-maintained green spaces (Larson et al. 2017).

The objectives of this study were to determine if golf course wildflower habitats and their relative floral richness affect the abundance and richness of bees and insect natural enemies, and the biological control of insect plant pests. To quantify these benefits on golf courses, we created wildflower habitats at two levels of species richness and quantified bees, natural enemies, and biological control compared to conventionally maintained turfgrass areas over one growing season. We predicted that, compared to maintained turfgrass, increasing floral abundance and richness would increase 1) bee abundance, richness, and diversity, 2) natural enemy abundance, richness, and diversity, and 3) the biological control of a key golf course insect pest.

Materials and methods

Study sites

This study was conducted on three golf courses in north-central Florida, USA: one in Gainesville, FL (29.648116, −82.368279) and two in Ocala, FL (29.264462, −82.146449; 29.105264, −82.261248). This region has a subtropical climate with average annual temperature and precipitation of 20.4 °C and 120.4 cm, respectively. Each course was chosen because it was representative of the average size, habitat type, and maintenance intensity of a southeastern U.S. golf course. On each course, we selected three 446 m^2 (24.38 m × 18.29 m) out-of-play turfgrass areas along fairway or tee margins to quantify arthropod communities. The distance from the edge of maintained turfgrass and the edge of our plots ranged from 0 to 2 m. Out-of-play areas on two courses were initially composed of *Paspalum notatum* Flugge (bahiagrass) maintained at a 10 cm mowing height, and areas on one course were composed of *Cynodon dactylon* (L.) Pers. (bermudagrass) maintained at a 5 cm mowing height. In all cases, prior to site selection, plots were regularly maintained, but not used for golf play.

Site preparation

On each course, we removed turfgrass from two of the 446 m^2 plots and left the third as a maintained turfgrass control. All plots were separated by at least 130 m. In October 2016, we prepared two plots on each course for seeding with wildflowers by making two applications of Fluazifop (Fusilade®, Syngenta Crop Protection, LLC) at 0.88 and 0.58 ml/ha, respectively, at a two-week interval. Two weeks after the final herbicide application, we used steel rakes to remove all dead plant material and expose bare soil. Due to a dense mat of dead *C. dactylon* plant material on one course, we used a sod cutter to physically remove the top 5 cm of dead plant material and expose bare soil in both plots.

To reduce risk of facilitating weed germination, we did not disturb the exposed soil in any plot. In November 2016, we broadcasted wildflower seed mixtures throughout each plot using a rotary push spreader at a rate of 1200 wildflower pure live seed (PLS) per 1 m^2, as determined by seed mass and species-specific PLS composition. Each seed mixture within a plot contained an equal proportion of PLS per species to ensure equal initial species representation. Control plots were left as maintained turfgrass under the previous management program at that site. *Paspalum notatum* plots were mown at a weekly interval and *C. dactylon* plots were mown twice per week from May through September.

Wildflower seed mixtures

We created two seed mixture treatments, one composed of five wildflower species (low diversity) and the second composed of nine wildflower species (high diversity) (Table 1). Floral availability over an entire growing season supports the greatest diversity of pollinators and other beneficial insects in agricultural landscapes (Williams et al. 2015). Therefore, species selection ensured varied bloom seasons to provide continuous floral resources March through December. The low diversity mixture was composed of *Coreopsis lanceolata* (lanceleaf coreopsis)*, Gaillardia pulchella* (Indian blanket)*, Spermacoce verticillata* (shrubby false buttonweed)*, Liatris gracillis* (slender blazing star)*,* and *Solidago stricta* (wand goldenrod)*.* The high diversity mixture was composed of the same species in the low diversity mixture as well as *Linaria canadensis* (Canada toadflax)*, Coreopsis basalis* (goldenmane tickseed)*, Monarda punctata* (spotted beebalm)*,* and *Conoclinium coelestinum* (blue mistflower).

Floral cover and composition

To quantify the temporal and spatial availability of floral resources, and confirm that our wildflower diversity treatments were distinct, we estimated percent floral cover and quantified species richness per plot every two weeks from April 2017 through September 2017. On each date, two individuals visually and independently estimated the percentage of each plot covered by flowering plants by dividing it into four quadrants, estimating the percent cover of each quadrant, and averaging them across the entire plot. On the same date, each flowering plant was identified to species, including planted wildflowers and volunteer species. Using these data, we calculated mean species richness and percent plot floral cover every two weeks for the entire survey period to determine if wildflower plots were providing sustained floral resources and if there were differences between wildflower mixture treatments.

Pollinator conservation

To quantify flying pollinating insects, we deployed colored pan traps at the centroid of each plot, 9 m and 12 m from the nearest edges (Fig. 1) (Leong and Thorp 1999). Blue, white, and yellow plastic bowls (12-oz Party City Holdco Inc) filled 1/3 with a 20% propylene glycol-soapy water solution were mounted to a corrugated plastic board adjusted to vegetation height (Campbell et al. 2017). Once per month, March 2017 through September 2017, pan traps were deployed. Pan trap contents were collected seven days later, pooled per plot and transferred to 50 ml centrifuge tubes (Falcon®) containing 70% ethanol. Collected bees (Anthophila) were identified to genus under a stereomicroscope in the laboratory using taxon-specific keys (Ascher and Pickering 2018).

Nesting sites are important for establishing pollinator communities and can be indicators of local community composition (Kremen et al. 2007). To quantify pollinator establishment, we surveyed solitary, cavity-nesting pollinators by also placing a nesting trap at the centroid of each plot (Fig. 1) (Campbell et al. 2017). Each trap was a 20.3 cm plywood cube with a 10.2 cm overhang on the front extending from the apex. We placed two 18.4 cm diameter PVC tubes inside each box, each containing approximately 44 bamboo reeds that were 18.4 cm long, sealed at a node, and ranged from 2.8 to 13.4 mm in diameter (Gathmann et al. 1994). In addition, each box contained four 5.1 × 10.2 × 18.4 cm wooden blocks drilled with two holes of each diameter: 5.6 mm, 6.4 mm, 9.5 mm, and 12.7 mm, to attract a variety of hymenopteran species (Campbell et al. 2017). Each trap was fixed to a wooden post and placed 1 m above the ground.

We checked each nesting box weekly July 2017 through October 2017 to quantify insect colonization. Any colonized cavities, as determined by capped openings, were collected from the nesting box and replaced with new reeds of a random size. We placed the open end of each colonized cavity into a 15 ml plastic vial sealed with Parafilm "M" laboratory film (Pechiney Plastic Packaging, Chicago, IL, USA) and kept them under an outdoor shelter where they were monitored for eclosion. All emerged insects were collected and identified to genus or species.

Table 1. Composition of two experimental wildflower species mixtures with estimated flowering periods indicated by white cells.

Low Diversity Mix				Season in Bloom			
Species	**Common name**	**Color**	**Annual / Perennial**	Spring	Summer	Fall	Winter
Coreopsis lanceolata	Lanceleaf coreopsis	Yellow	Perennial				
Gaillardia pulchella	Indian blanket	Red	Perennial				
Spermacoce verticillata	Shrubby false buttonweed	White	Perennial				
Liatris gracillis	Slender blazing star	Purple	Perennial				
Solidago stricta	Wand goldenrod	Yellow	Perennial				
High Diversity Mix				**Season in Bloom**			
Species	**Common name**	**Color**	**Annual / Perennial**	Spring	Summer	Fall	Winter
Linaria canadensis	Canada toadflax	Purple	Annual / Biennial				
Coreopsis lanceolata	Lanceleaf coreopsis	Yellow	Perennial				
Coreopsis basalis	Goldenmane tickseed	Yellow	Annual				
Gaillardia pulchella	Indian blanket	Red	Perennial				
Mondarda punctata	Spotted beebalm	Purple	Perennial				
Conoclinium coelestinum	Blue mistflower	Blue	Perennial				
Spermacoce verticillata	Shrubby false buttonweed	White	Perennial				
Liatris gracillis	Slender blazing star	Purple	Perennial				
Solidago stricta	Wand goldenrod	Yellow	Perennial				

Natural enemy conservation

To quantify the abundance and diversity of natural enemies visiting wildflower plots, we deployed three 7.6 × 12.7 cm yellow sticky cards (Olson Products, Medina, Ohio, USA) along the midline of each plot spaced 8 m apart (Fig. 1). Once per month, March 2017 through September 2017, we mounted each sticky card to a stake 0.75 m above the ground and collected it seven days later. Collected cards were placed into a clear, re-sealable plastic bag (Great Value ™, Bentonville, Arizona, USA) and taken to the laboratory where they were kept in a − 15.6 °C freezer. We identified all captured natural enemies to family using a stereomicroscope. Using these data, we quantified natural enemy abundance, family-level richness, and Simpson (1-D) diversity index for each wildflower diversity treatment (Simpson 1949).

Biological control services

Changes in natural enemy communities (i.e., increased predator abundance) may not directly translate to differences in the biological control services those organisms provide (Dobbs and Potter 2016; Gardiner et al. 2013). Therefore, to determine if wildflower presence or diversity affected the biological control of herbivorous plant pests, we deployed a sentinel pest experiment similar to Dobbs and Potter (2016) and Frank and Shrewsbury (2004). Along four transects adjacent to each plot, we quantified larval dissapearance rates associated with flying insect natural enemies of fall armyworm, *Spodoptera frugiperda* (Lepidoptera: Noctuidae). *Spodoptera frugiperda* is a common damaging pest of turfgrasses on golf courses and other urban turfgrass settings (Held and Potter 2012; Nagoshi and Meagher 2004).

Sentinel prey arenas were created using a cup, culinary basket, turfgrass plug, and greenhouse fabric staples with the goal of eliminating predation from ground-dwelling organisms. Specifically, we placed a 6 cm-diameter plug of *Zoysia matrella* cv. 'Diamond' turfgrass into a clear plastic cup (11.5 cm wide by 4.5 cm deep) (Fabri-Kal Corp., Kalamazoo, MI, U.S.A.), covered it with an upside-down 20 cm diameter culinary frying basket (1.5 cm square mesh openings), suspended the cup from the basket with twist ties (leaving 0.5 cm gap from the basket), and attached 25 cm ground cover anchoring pins to the sides of the culinary basket so that the bottom of the cup was suspended 15 cm above the ground (Fig. 1). Mesh wire culinary baskets excluded vertebrate predation (Dobbs and Potter 2016) and suspending the cup excluded predation from common turfgrass-dwelling predators like ants, beetles, and spiders (Held and Potter 2012). We wrapped each bottom half of the grass plug in plastic wrap (Great Value ™, Bentonville, Arizona,
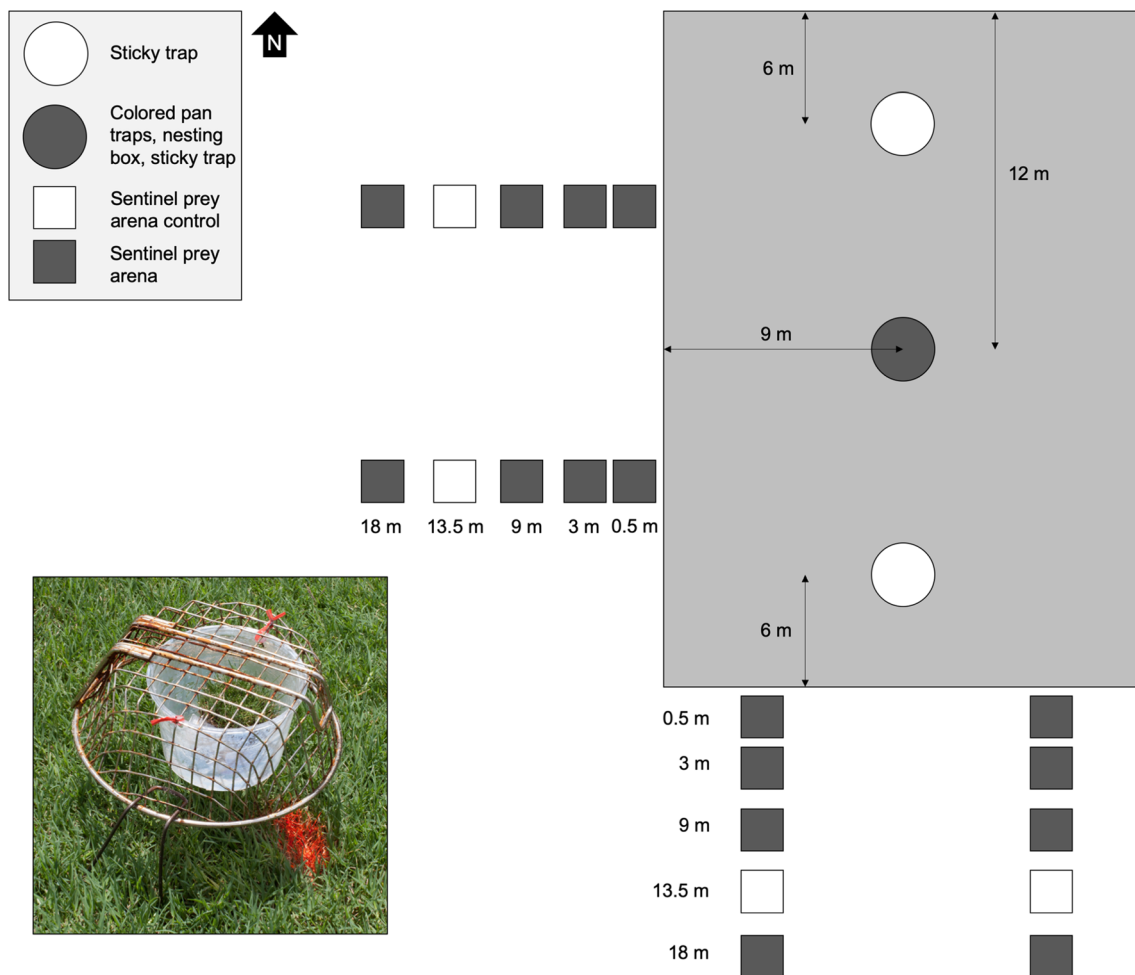
Fig. 1 Plot layout and sampling design used to quantify bee and insect natural enemy communities, and larval disappearance rates. Each plot was 446 m². Photo inset demonstrates the sentinel prey arenas used to quantify larval disappearance rates

USA) to reduce plant desiccation and prolong the food source for *S. frugiperda*.

Spodoptera frugiperda larvae were obtained from Benzon Research Co. (Carlisle, Pennsylvania, USA), where they were reared on a soy-flour and wheat germ base artificial diet. We placed four, third-instar larvae onto the *Z. matrella* plug within each arena. The inside lip of each cup was coated with a thin layer of petroleum jelly (Vaseline®, Unilever, Rotterdam, Netherlands) to prevent caterpillar escape. To quantify larval escape rates and ensure our measure of biological control was due to predation, we also deployed control arenas created as described above, but with the culinary basket covered by organza fabric (Casa Collection, Joann, Hudson, OH, USA) to prevent access to flying natural enemies, but allow larvae to escape from the cup if able.

Sixteen functional arenas and four control arenas were placed along four transects extending from each plot in June and July 2017 (Fig. 1). We placed arenas along two transects on the north and west side of each plot with five arenas per row and 20 arenas per plot. Functional arenas were placed 1, 3, 9,

and 18 m from the edge of the plot. All control arenas were located 13.5 m from the plot edge. We collected the arenas 36 h after deployment when remaining larvae were predominantly fourth instar to quantify predation and parasitism rates. We recorded the number of larvae remaining per cup and whether they were dead or alive. Live caterpillars were individually placed on artificial diet in 59 ml Dixie cups ® (Georgia-Pacific Consumer Products LP, Atlanta, Georgia, USA) and maintained at 27 °C and 50% RH where they were monitored daily for parasitism until they eclosed as adult moths.

Statistical analysis

All statistical analyses were performed in JMP Pro Version 13.1 (SAS Institute Inc., Cary, North Carolina). To evaluate the effects of our plant diversity treatments (high wildflower, low wildflower, turfgrass control) and survey date on floral richness and percent plot floral cover, we used mixed-effects analysis of variance, treating golf course as a random effect.

Models initially included the survey date x plant diversity interaction term, but this term was removed due to no significant interaction. Insect pollinators and natural enemies were surveyed once per month for seven consecutive months. We took the mean of these surveys of abundance and richness across all seven dates for each golf course, averaged values across golf courses, and compared plant diversity treatment means ($N = 3$ for each treatment). To determine the effect of plant diversity treatments on bee and natural enemy abundance, richness, and Simpson (1-D) diversity, we used mixed-effects analysis of variance, treating golf course as a random effect. Finally, we used a mixed-effects model to evaluate the main effect of plant diversity treatment and distance from plot edge on biological control rates for each survey date separately (June and July), treating golf course as a random effect. Response variable means were compared between plant diversity treatments using Tukey's HSD means comparisons. Model effects were considered significant if $P < 0.05$.

Results

Floral cover and composition

Bi-weekly plot evaluations indicated that we successfully created dense stands of flowering plants that were in some level of bloom during the entire survey period, 19 April 2017 through 14 September 2017. Percent plot floral cover averaged over 90% for both wildflower treatments throughout the study, while floral cover in the turfgrass control plots ranged from over 90% during one week in early June (due to weeds) to less than 5% by mid-July (Fig. 2a). Importantly, high diversity and low diversity wildflower plots had significantly greater percent floral plot cover than the turfgrass plots through the duration of our survey period ($F_{2,79.6} = 99.1$, $P < 0.0001$). There was a nearly significant effect of date ($F_{11,76.2} = 1.83$, $P = 0.06$) on percent plot cover, which was most apparent in the turfgrass plots due to flowering weed presence in the early months of the study (Fig. 2a). Wildflower phenology transitioned between species as expected over the year, while providing continuous floral resources. In the low diversity plots, *C. lanceolata* averaged approximately 20% plot cover April through June. *Gaillardia pulchella* dominated both wildflower mixture treatments and was the most abundant species in the high diversity plots June through August, and through the entire survey period in the low diversity plots. *Coreopsis basalis* and *C. lanceolata* were the dominant species in the high diversity plots April through May and *M. punctata* became the dominant species in September. Volunteer flowering species included *Richardia spp.* (Mexican clover)*, Verbena spp., Oenothera laciniata* (evening primrose)*,* and *Conyza canadensis* (Canadian horseweed).
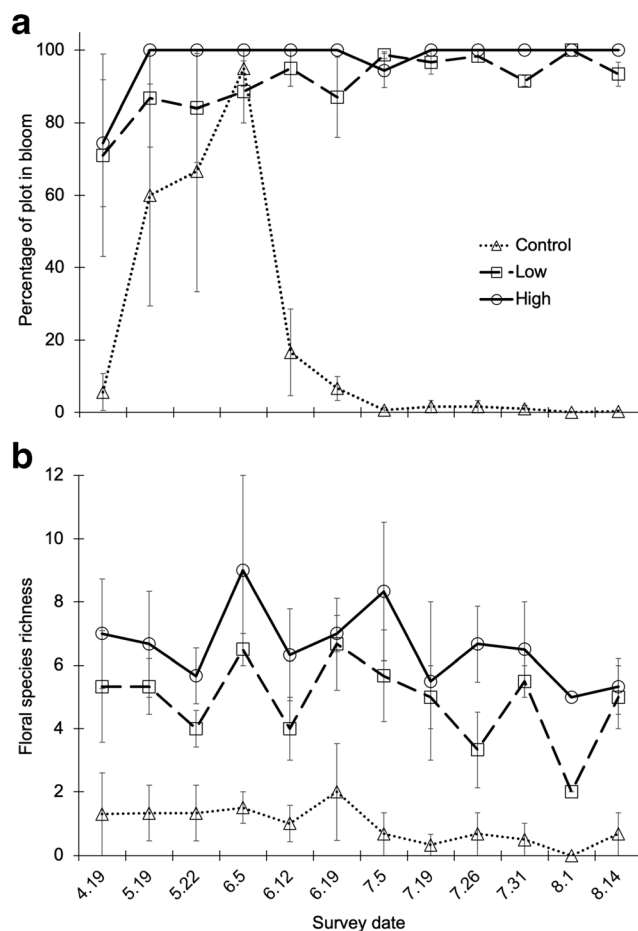

Fig. 2 The bi-weekly average (A) percentage of each experimental plot that was occupied with flowering plants and (B) number of different flowering plant species in bloom within each experimental plot for each treatment: turfgrass control, low diversity wildflower mixtures, and high diversity wildflower mixtures from April through August 2017. Error bars represent standard error

In support of our intended habitat manipulation, we created wildflower plots that differed in floral species richness. On average, the high diversity wildflower plots had 3 to 12 species in bloom with a mean ± SE of 6.65 ± 0.42, the low diversity plots had 1 to 9 species in bloom with a mean of 4.97 ± 0.34, and the turfgrass plots had 0 to 5 species in bloom with a mean of 1 ± 0.23. Floral species richness was significantly different between treatments such that each was statistically distinct (Fig. 2b) ($F_{2,79} = 124.4$, $P < 0.0001$). Date was not a significant factor in predicting average floral richness ($F_{11,79} = 1.5$, $P = 0.17$).

Pollinator conservation

Among surveyed pollinators, we collected native bees from 13 genera within four families (Megachilidae, Apidae, Andrenidae, Halictidae) (Table S1). Bees were affected by wildflower species richness such that the high diversity wildflower plots attracted significantly more bees than the low diversity and turfgrass plots (Table 2) (Fig. 3a). Interestingly,

Table 2 The average abundance, richness, and diversity of bees and natural enemies captured using pan traps, yellow sticky cards, and nesting boxes within turfgrass, low diversity wildflower, and high diversity wildflower plots on golf courses in 2017. Native bees identified to genus and natural enemies and nesting box colonizers identified to family. Means not followed by the same letter differ significantly using Tukey's HSD means comparison

	Plant diversity treatments	Mean	SE	$F_{2,3}$	P
Native bees					
Abundance	Turfgrass	6.5 b	1.3	20.3	**0.008**
	Low Diversity WF	8.9 b	2.1		
	High Diversity WF	27.6 a	3.9		
Richness	Turfgrass	1.4	0.1	1.9	0.257
	Low Diversity WF	2.8	0.8		
	High Diversity WF	2.2	0.3		
Simpson's Diversity (1-D) Index	Turfgrass	0.55 ab	0.07	4.5	**0.044**
	Low Diversity WF	0.6 a	0.02		
	High Diversity WF	0.4 b	0.03		
Natural enemies					
Abundance	Turfgrass	14.3 b	6.0	26.8	**0.005**
	Low Diversity WF	41.3 b	14.4		
	High Diversity WF	75.7 a	14.6		
Richness	Turfgrass	2.9	0.4	0.3	0.731
	Low Diversity WF	3.3	0.4		
	High Diversity WF	3.5	0.4		
Simpson's Diversity (1-D) Index	Turfgrass	0.5	0.2	1.7	0.301
	Low Diversity WF	0.5	0.1		
	High Diversity WF	0.4	0.1		
Nesting box colonizers					
Abundance	Turfgrass	32	20.3	0.65	0.572
	Low Diversity WF	11.7	6.5		
	High Diversity WF	16	8.1		
Richness	Turfgrass	4	1.5	1.0	0.444
	Low Diversity WF	2.7	0.9		
	High Diversity WF	2.0	0.6		

Simpson's diversity for bees was greater in the low diversity wildflower plots compared to the high diversity and turfgrass plots (Table 2). This is likely because native bee communities within the high diversity wildflower plots were dominated by *Halictus poeyi* and *Lasioglossum* spp. (Table S1), which may have preferred the additional plant species in those pots. We found no effect of wildflower presence or diversity on native bee genus-level richness.

Our pollinator nesting boxes were colonized by several insect groups that were not collected in our pan traps. Over the duration of our survey period, insects from at least eight genera within six families and three orders colonized the nesting boxes, primarily parasitic and predatory species. Surprisingly, the only bees to colonize our boxes were *Megachile spp*. (Hymenoptera: Megachilidae). Remaining nesting box colonizers were parasites of other nesting box colonizers (flies in the subfamily Sarcophaginae, *Toxophora spp.* (Diptera: Bombyliidae), *Macrosiagon spp.* (Coleoptera: Ripiphoridae), and wasps in the family Chrysididae) or predators of insect herbivores (potter wasps). Interestingly, 82% of nesting box colonizers were potter wasps (Hymenoptera: Vespidae: Eumeninae: *Euodynerus spp*., *Pachodynerus erynnis*, and *Parancistrocerus spp.*), which were the most abundant nesting box colonizers across all treatments. Moreover, nesting boxes in the turfgrass plots numerically supported the greatest abundance of insects, although not significantly ($P > 0.44$) (Table 2). Among nesting box colonizers, *P. erynnis* was seven times more abundant than the next most common colonizer. This is important because *P. erynnis* is a specialist predator of Lepidopteran larvae, particularly *S. frugiperda*, a key insect pest of golf course turfgrasses (Held and Potter 2012).

Natural enemy conservation

Coinciding with our pollinator surveys, we deployed yellow sticky cards within each plot to passively monitor flying insect
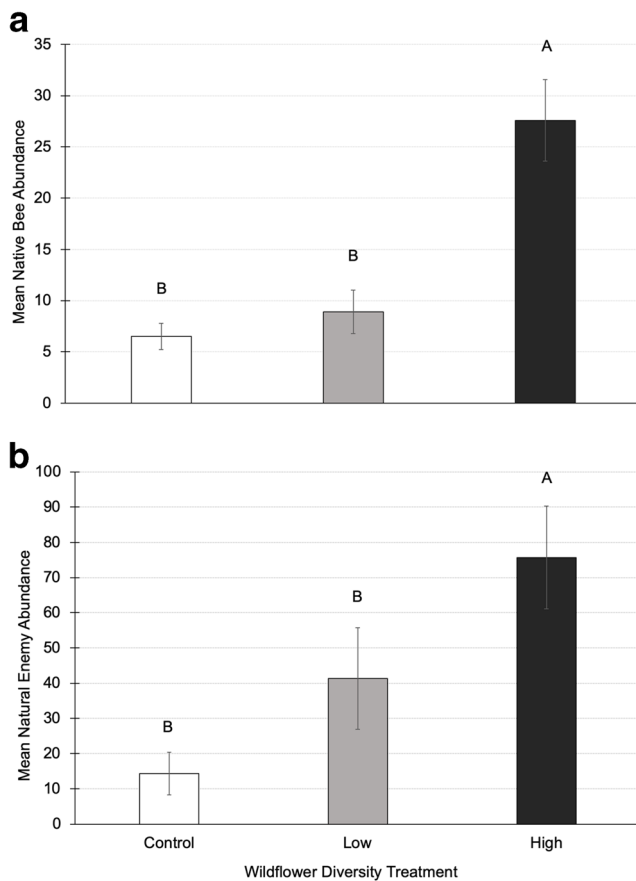
Fig. 3 Monthly, 7-day average (A) native bee abundance and (B) insect natural enemy abundance per wildflower diversity treatment from March 2017 through September 2017 using colored pan traps and yellow sticky cards, respectively. Each bar represents abundance averaged across all golf courses ($n = 3$). Error bars represent standard error and different letters above bars indicate statistical differences ($P < 0.05$)

natural enemies. Sticky cards captured natural enemies from 13 different families over the duration of our survey period, predominantly composed of two families of generalist predators, Dolichopodidae (Diptera) and Anthocoridae (Hemiptera) (Table S1). We did not detect any effect of wildflower presence or richness on natural enemy family-level richness or Simpson's Diversity. Compared to the turfgrass plots, mean monthly natural enemy abundance was approximately five and three times greater in the high and low diversity wildflower plots, respectively. Increasing wildflower species richness increased natural enemy abundance, but only significantly in the high diversity wildflower plots (Table 2) (Fig. 3b).

Biological control services

Our estimates of biological control of *S. frugiperda* adjacent to study plots were high across all sites, averaging over 50%. On average, 36% of *S. frugiperda* larvae disappeared from our "control" sentinel prey arenas for unknown reasons. This may be explained by larval escape or cannibalism. Larval

disappearance from the control arenas was not different between plant diversity treatments or plots on either date ($P > 0.25$). Therefore, we compared our measures of predation across all sites without adjusting for unexplained larval loss. Similarly to Dobbs and Potter (2016), we did not find any larval parasitism at any locations. Therefore, our measure of biological control is more representative of predation from flying arthropods. On both dates, disappearance rates adjacent to the high diversity wildflower plots were greater than around turfgrass plots. We found that on average, creating wildflower habitats translated to a nearly 50% increase in the disappearance of *S. frugiperda* larvae in managed turfgrass areas adjacent to wildflower plots compared to those adjacent to turfgrass plots (Fig. 4). There was an effect of wildflower species richness on larval disappearance in June, such that high diversity plots experienced significantly greater disappearance rates than the turfgrass plots ($F_{2,138} = 6.02$, $P = 0.0031$) (Fig. 4). In July, both wildflower diversity treatments experienced significantly greater disappearance rates than the turfgrass plots ($F_{2,134.1} = 8.23$, $P = 0.0004$) (Fig. 4). There was no effect of distance from the study plots (up to 18 m) on larval disappearance rates in June ($F_{3,129} = 0.22$, $P = 0.88$) nor July ($F_{3,125} = 0.40$, $P = 0.75$), suggesting that benefits of wildflower plots extend at least 18 m from the plot edge.

Discussion

Pollinator conservation is increasingly important as urbanization and other anthropogenic factors cause declines in pollinator communities around the globe (Goulson et al. 2015). Since urban ecosystems can support relatively diverse bee communities (Hall et al. 2017), but are characterized by highly disturbed plant communities and altered abiotic conditions,
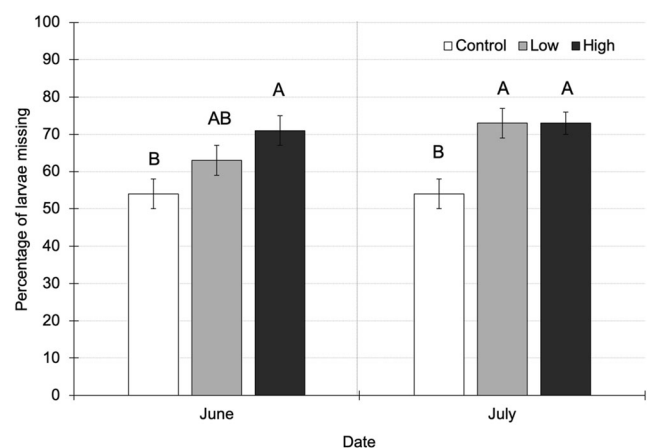

Fig. 4 Mean percentage of *S. frugiperda* missing after 36 h in maintained turfgrass areas adjacent to our experimental treatments: turfgrass, low diversity wildflower mixture, and high diversity wildflower mixture. Different letters above bars indicate statistical differences (P < 0.05) within a survey date (June and July). Error bars represent standard error

large continuously vegetated green spaces within them can be highly valuable. As we predicted, replacing maintained turfgrasses with flowering plants increased bee and insect natural enemy abundance on golf courses and the strength of this effect depended on floral species richness. Our results demonstrate that creating relatively large (446 m^2) and diverse floral habitats on golf courses provides multiple benefits, conserving insect pollinators and increasing biological control of an important pest species. Since golf courses are often highly maintained to protect plant quality and turfgrass playability, a 50% increase in biological control of a turfgrass pest may translate to direct environmental and economic benefits. With over 1100 golf course facilities in Florida (Hodges and Stevens 2010) and about 16,000 in the U.S. (Held and Potter 2012), such conservation efforts would represent a significant footprint of pollinator conservation habitat and reduced management inputs within urban ecosystems.

It is important to recognize that not all wildflower habitats are equal in function. For example, although both wildflower diversity levels averaged over 90% bloom coverage for the duration of the study, mixtures of nine wildflower species supported a three-fold increase in native bees compared to mixtures of five species. Across all wildflower plantings, *G. pulchella* was the most abundant species in both mixtures throughout the survey period. This suggests that the additional species in the more diverse plantings played an important role in supporting native bees, particularly *C. basalis*, *C. coelestinum*, and *M. punctata*. Similarly, Jha and Kremen (2013) found that floral species richness was the best predictor of foraging distance for bumblebees across agricultural and human-dominated landscapes. This emphasizes the importance and value of floral richness for native bees, which often specialize on specific plant species or genera (Potts et al. 2003). Therefore, uninformed wildflower plantings likely increase pollinator abundance, but proper plant selection based on plant species phenology and richness are important for maximizing bee conservation.

Although our pan trap surveys indicated an effect of wildflower richness on pollinator abundance, our nesting traps did not. Moreover, our nesting traps were dominated by predatory wasps instead of bees. In addition to providing appropriate floral resources, suitable nesting habitat is critical to supporting solitary-nesting pollinators (Potts et al. 2005). The majority of bee species, including those in Florida, nest in the ground (Pascarella and Hall 2019). Therefore, our pan traps are likely a better tool than nesting boxes at capturing the effect of our wildflower plots on bee conservation. Although not significant, the nesting traps in our turfgrass control plots numerically supported the most insects and highest family-level richness, suggesting that providing nesting habitat in and of itself has conservation value within open turfgrass areas. Higher abundance and richness may be explained by the presence of more open ground and an ability to forage for prey within some of our turfgrass control plots compared to dense wildflower plantings (Ballare et al. 2019; Potts et al. 2005). Among the insect groups that colonized the nesting traps, *P. erynnis* was seven times more abundant than the next most abundant species. *Pachodynerus erynnis* is a solitary wasp that builds nests in cavities and attacks caterpillars in the families Amphisbatidae, Elachistidae, Erebidae, Coleophoridae, Noctuidae, Oecophoridae, Pyralidae, and Torticidae (Carpenter 1986; Krombein 1967), several of which contain pest species of golf course plants (Held and Potter 2012; Potter 1998). More pertinently, *P. erynnis* is a predator of fall armyworm, *S. frugiperda*, which is a key pest of golf course turfgrasses around the world (Potter 1998). Although we did not quantify the biological control services provided by this species, we did observe it capturing *S. frugiperda* larvae and provisioning its nests with them at our study sites. Despite this predator being more abundant in turfgrass nesting boxes, our larval disappearance data suggest that overall foraging for prey was higher in areas adjacent to wildflower plantings.

In general, flying insect natural enemies became more abundant when we replaced turfgrass areas with wildflower plantings. This is not surprising since areas of relatively complex vegetation (i.e., structurally or florally) are predicted to support more abundant and diverse natural enemies (Bottrell et al. 1998; Tscharntke et al. 2007). Although we saw an increase in natural enemy abundance with increasing wildflower richness, we did not see an effect on family-level richness. This may be because family level is too coarse resolution, although other work suggests family-level is good to moderate at predicting insect species richness (Baldi 2003; Mazón 2015; Vieira et al. 2012). Despite not finding a consistent effect of plant species richness on biological control rates, there was a numerical increase with increasing plant richness, which may have become more detectable with an higher level of plant species richness (Blaauw and Isaacs 2015; Scheid et al. 2011). Regardless of plant species richness, our sentinel prey experiment suggests that wildflower presence increases biological control compared to turfgrass plantings. This is in contrast to a few other studies (Braman et al. 2002; Dobbs and Potter 2016), who found that predation was no different between turfgrass areas adjacent to other turfgrass or adjacent to conservation plantings. This is likely because previous studies did not exclude ants or other ground-dwelling predators, which are the most ubiquitous and active turfgrass predators (Dobbs and Potter 2016). We excluded ground-dwelling predators in our sentinel prey experiment, which allowed us to isolate flying natural enemy predation. In addition, we designed our plantings to attract flying natural enemies based on previous literature (Portman et al. 2010), who like us, found increased biological control rates adjacent to floral habitats.

Although we excluded ground-dwelling natural enemies in our study, they are an important component of wildflower habitats that occur beneath the floral resources (López and Potter 2000). Plant architectural complexity and density can provide refugia that harbor ground-dwelling predators like beetles and spiders which provide important pest control services in turfgrass systems (Dobbs and Potter 2016; Frank and Shrewsbury 2004). For example, Frank and Shrewsbury (2004) found that adding conservation strips along golf course fairway margins augmented predatory beetle populations and caterpillar pest control up to 10 m from the conservation planting. Since we did not quantify ground-dwelling arthropods or their services in this study, our estimates of arthropod conservation and biological control associated with our wildflower habitats are conservative. Future work should investigate ancillary effects of wildflower species richness and other metrics of pollinator conservation habitats on ground-dwelling arthropod communities and their services.

Golf course wildflower plantings present an opportunity to merge societal, conservation, and economic interests (Colding et al. 2006; Dobbs and Potter 2015). In addition to conserving insects and promoting pest control, converting areas of maintained turfgrass to wildflowers reduces water, fuel, chemical, and labor inputs (Gross and Eckenrode 2012; Lyman et al. 2007). Each of these conservation benefits can help mitigate negative environmental impacts of golf course management and urbanization, while also providing educational opportunities for the general public (Larson et al. 2017; Dobbs and Potter 2015). Although golf course pollinator conservation efforts have been underway for several years (Dobbs and Potter 2015), this study highlights the importance of creating evidence-based plantings by illustrating the value of plant species richness. Plant species richness has also been employed in agricultural systems to reduce insect abundance. Increasing plant diversity and reducing the size of monoculture plantings is a well-established approach for reducing herbivore pressure, particularly specialists (Tahvanainen and Root 1972). Thus, increasing plant species richness may negate conservation efforts for some insects of conservation concern, like monarchs (*Danaus plexipus*), which specialize on *Asclepias spp.* hosts. Future work should investigate strategies for creating golf course conservation habitats that target specialist insects. Other factors should also be explored to further inform optimal conservation strategies in these urban green spaces. For example, it is poorly understood how golf course landscape context or the design of conservation plantings within a golf course may affect the conservation value of that space. Strategically located plantings within a golf course may amplify conservation efforts by developing corridors (Resasco 2019) or by taking advantage of adjacent or surrounding landscapes (Goddard et al. 2010; Tscharntke et al. 2007; Burkman and Gardiner 2014). Finally, since golf courses are largely recreational and aesthetic, it is important that these habitats

be strategically designed to maximize aesthetic quality, course functionality, and habitat longevity, each of which require additional research to facilitate widespread implementation. As urbanization progresses, such strategies to integrate wildlife and resource conservation with anthropogenic activities are critical steps towards increased sustainability.

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