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Flower plantings promote insect pollinator abundance and wild bee richness in Canadian agricultural landscapes

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Abstract
Global declines in pollinator populations are an ongoing concern from biodiversity and food security viewpoints. A growing conservation initiative in agricultural landscapes is the establishment of wildflowers on marginal lands to provide floral resources and habitat for pollinators. However, the effectiveness of such conservation and restoration efforts are not always assessed. We assessed the effectiveness of a private sector pollinator conservation initiative by (1) comparing insect abundance and richness between planted flower plots and control plots and (2) assessing changes between years. Over two years, planted flower plots and control plots (i.e. out-of-production farm areas) located in Canada were surveyed for insects using visual observation, netting, and pan trapping methods. Significantly more pollinators, especially wild bees, and higher wild bee richness were found in planted plots than control plots. Plot size had no effect on insect abundance and richness indicating that even small-scale flower plantings can provide benefits to pollinator communities. While pollinator, predator, and herbivore arthropod abundance and richness were stable or declined between years, likely due to adverse weather conditions in the second year of the study, wild bee abundance and richness increased over the same period. Our results support that flower plantings can be a successful conservation tool to increase pollinator and wild bee abundance and biodiversity within agricultural landscapes.

Implications for insect conservation Small-scale flower plantings within agricultural landscapes are a simple and effective conservation management strategy to support local insect pollinator populations.

Keywords Biodiversity · Native bees · Habitat enhancement · Conservation · Floral resources

Introduction

Pollinating insects are a vital part of natural ecosystems with over 80% of flowering plants dependent on pollinators to reproduce (Ollerton et al. 2011). Ongoing declines in wild pollinator populations have dire implications for the preservation of natural biodiversity (Burkle et al. 2017). Native insect pollinator populations have been negatively impacted by multiple factors including habitat degradation and destruction (Kearns et al. 1998; Potts et al. 2010; Durant

& Otto 2019), the introduction of invasive species or disease (Potts et al. 2010; Cameron et al. 2011; Ollerton 2017; Cameron & Sadd 2020), and the adverse effects of agricultural activities (Kearns et al. 1998; Ollerton 2017; Cameron & Sadd 2020). Biodiversity is directly impacted when declines in pollinators remove species from the landscape, and there are also serious concerns for biodiversity impacts to cascade throughout ecological communities if wild plant pollination services are disrupted (Vasiliev & Greenwood 2020).

Insect pollinators also play an important role in human food production, with 35% of global agriculture relying to some degree on animal pollination services to be productive (Klein et al. 2007). Declines in natural and managed pollinators puts the security of human food production in question (Burkle et al. 2017) and declines in crop production are already apparent in some regions due to deficient rates of pollination (Cusser et al. 2016; Koh et al. 2016; Monasterolo et al. 2015; Reilly et al. 2020). Like native pollinators, managed bee colonies have been degraded by a number

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of threats, including colony collapse disorder (vanEngelsdorp et al. 2009; Gray et al. 2020), changes in environment (Potts et al. 2010; Durant & Otto 2019; Flores et al. 2019), and introduction of novel parasites or disease (Potts et al. 2010; Cornman et al. 2012; McMenamin & Genersch 2015; Simon-Delso et al. 2016). These issues underscore the continued need to improve our understanding of pollinator populations across diverse communities and for the effective monitoring and conservation of populations across temporal and spatial scales (Burkle et al. 2017; Crossley et al. 2020; Potts et al. 2010).

The need for pollinator-supportive management practices and diversification of agricultural landscape extends from the fundamental connection between pollinating insects and the ecological function they provide. While agricultural areas are generally poor habitat for native insect pollinators due to intensive habitat disturbance and pest control measures, on-farm management to improve floral resources and habitat (e.g. nesting or overwintering sites) can restore and enhance pollinator populations (M'Gonigle et al. 2015; Grass et al. 2016; Burkle et al. 2017; Ollerton 2017; Campbell et al. 2019; Chatterjee et al. 2020). Even small-scale restoration efforts within agricultural landscapes can improve pollinator diversity, including support for rare or more specialized pollinator species (Kremen & M'Gonigle 2015). In addition to improving pollinator abundance, increasing the diversity of pollinator communities is important for improving community resilience to changes in the environment and to stochastic events (Burkle et al. 2017; Ollerton 2017). For example, bees are unable to forage during cold and rainy weather whereas other pollinators are still active and providing pollination services (Vasiliev & Greenwood 2020). Similarly, diverse native populations of pollinators can provide supplemental crop pollination services when sufficient managed populations are not available, buffering detrimental effects resulting from stochastic events such as colony collapse (Burkle et al. 2017; Vasiliev & Greenwood 2020). Therefore, it is important to consider both native pollinator abundance and diversity during conservation of these communities.

Conservation and restoration initiatives aimed at improving pollinator habitat can vary greatly by region depending on social, political, and legal structures (Vasiliev & Greenwood 2020). In addition to programs run by government or not-for-profit organizations, private corporations can be well placed to lead initiatives to improve or restore biodiversity (Addison et al. 2018; Smith et al. 2019), though the effectiveness of these conservation actions is not always assessed (Sayer et al. 2017). An example of a corporate initiative to improve pollinator biodiversity is Operation Pollinator, an international program run by Syngenta, a global agribusiness that produces crop protection solutions and seeds (https://www.syngenta.com/en/sustainability/operation-pollinator).

This program is intended to provide knowledge, expertise, and resources designed to encourage and support the creation of essential habitat and food sources for pollinating insects within a variety of landscapes.

In Canada, under a multi-year pilot program, Operation Pollinator assisted farmers across Alberta (AB), Saskatchewan (SK), and Manitoba (MB) with efforts to enhance insect pollinator habitat and biodiversity. Volunteer farmers were provided with a seed mix free of charge, tailored to provide floral resources and create pollinator habitat. Each farmer was expected to convert 0.4–1.2 ha (1–3 acres) of marginal farmland to non-farmed pollinator habitat. Areas recommended for conversion included lower-productivity lands (not too saline or sensitive to drought), areas adjacent to hedgerows and other ecologically sensitive areas (e.g., ponds, drainage ditches, other riparian areas), cut-outs such as the corners of fields, or generally any site with decent drainage, adequate soil fertility, and suitable sun exposure (Syngenta Group Company 2017). While survey feedback from participating farmers has been positive (J.E.L. *unpubl. data*), the impact of this pilot program on local pollinator communities has yet to be quantified. Here, we compare insect abundance and richness between sites enhanced with plantings of floral resources to neighbouring out-of-production areas to assess the efficacy of this approach to restore and enhance native pollinator communities within agricultural landscapes.

Methods

Study areas

In September 2017, Syngenta provided 100 farmers across Alberta, Saskatchewan, and Manitoba with a commercially available flowering plant seed mix for planting. A subset of these farmers was selected for this study on a volunteer basis (five from each province). The flowering plant seed mixture provided was consistent across participants and contained approximately 10% Alsike clover (*Trifolium hybridum*), 20% Bird's-foot trefoil (*Lotus corniculatus*), 20% Phacelia (*Phacelia sp.*), 20% Red clover (*Trifolium pretense*), 10% Timothy (*Phleum pratense*), and 20% Yellow (*Melilotus officinalis*) and/or White (*Melilotus alba*) sweet-clover. The seed mixture was selected as a high-quality, robust mix that would provide a succession of flowering plants to deliver a continuous source of pollen and nectar for pollinators, and to be conducive to growing across the geographic range of the program. The mixture was selected with a focus on the provision of high-quality nutrition, including the protein-dense pollen of legumes, not with the objective of restoring native plant communities. Due to variations in the environmental characteristics between farms, differing results were found

in the success of the seed mixture propagation in 2018. In 2019, additional seeds were shipped to project participants in Alberta after poor seed germination was reported. Though vegetation surveys were not conducted during this pilot project, in general, flower plots were well established across the three provinces by 2019 (NRSI 2019; Fig. S1).

Treatment and control plots were situated in northern Alberta (along the Peace River Valley), in southern Saskatchewan, and in southern Manitoba (Fig. 1). In addition to the 15 treatment (seed mix planted) plots, in 2018 one control plot (an unseeded, out-of-production area) was established in each province for a total of 18 sites. In 2019, four more control plots were established in each province bringing the total number of sites to 30 (15 treatment + 15 control plots). Each control plot was located on the same farm as each treated plot and was selected to represent the typical out-of-production areas on the farms involved. These areas generally consisted of fallow field areas, flowering cover crops (e.g., clovers), or areas of sparsely treed field. Plots ranged in size from 0.13 to 1.70 ha (0.3 to 4.2 acres) averaging 0.65 ha (1.6 acres) in size. Given variation in the size of plots, we included size in our statistical models (details below).

Survey methodology

To sample insect species, a combination of visual surveys, sweep netting, and pan trapping were conducted at each study site in 2018 and 2019. These methods are commonly used to sample insects in a wide variety of different habitat types (O'Connor et al. 2018). Visual surveys allow an observer to document relatively conspicuous species such as butterflies and bumble bees. Sweep netting is relatively labour intensive but is an effective method for quickly sampling a wide range of insects in a short amount of time. Coloured bowls used in pan trapping mimic flowers and are effective at capturing many species of bees, some lepidoptera, flower-visiting flies and other insects. For the purposes of this project, a combination of these sampling techniques was employed to provide a representative sample of the insect community present within the research plots.

Visual surveys were conducted monthly at each site from June to September in both 2018 and 2019. Each survey consisted of a 30-min visual search along a random wandering transect. During the timed survey, visually identified pollinating insects were recorded with a focus on conspicuous species such as day flying moths, butterflies, and bumble bees. A net was used to capture individuals for identification purposes when necessary. During passive visual surveys for butterflies, bumble bees and other easily identifiable specimen, species names were recorded in the field. In some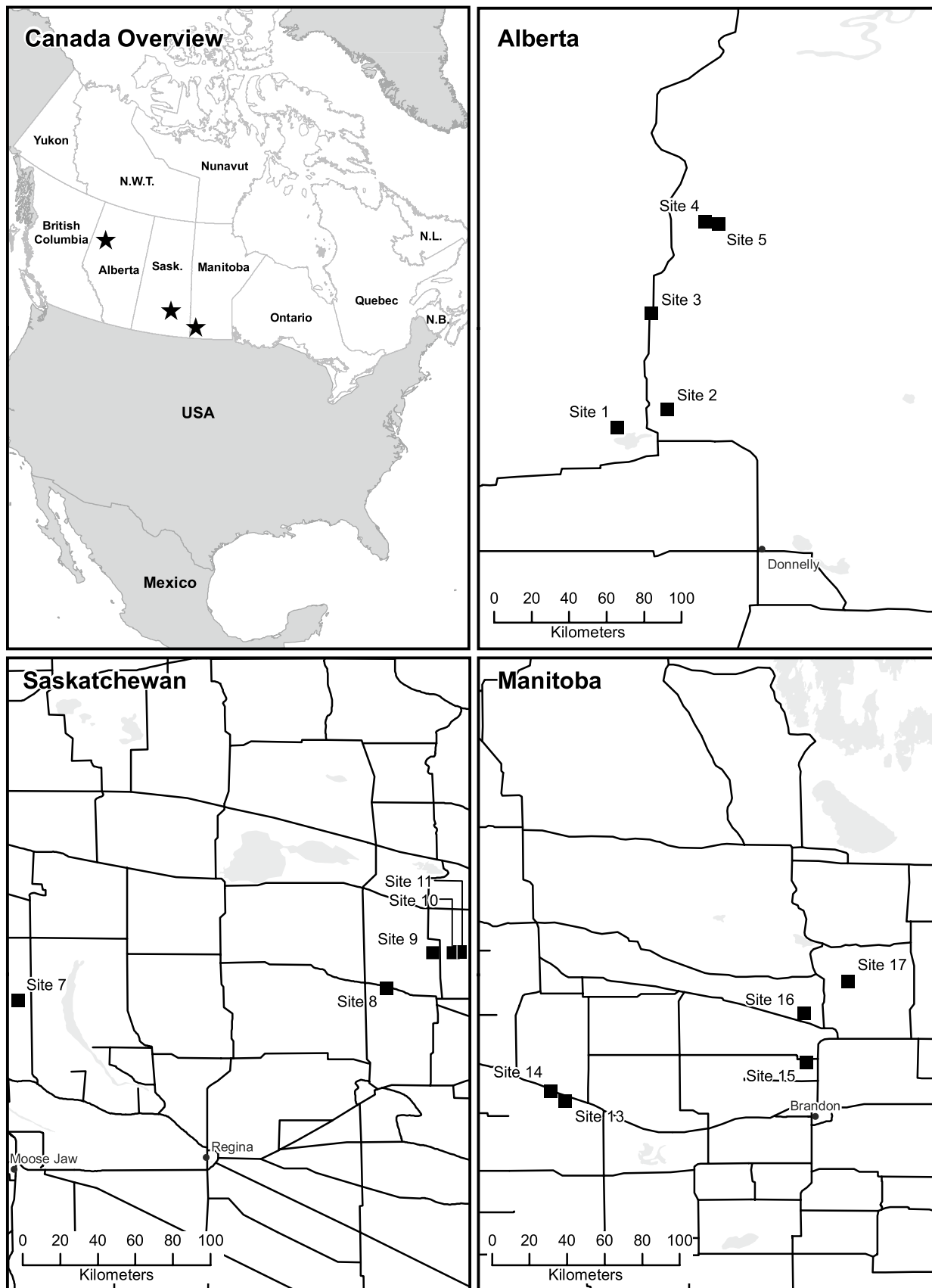 cases, observed bees and butterflies evaded capture and were not identified to species. Where possible, captured individuals were identified in the field and released.

Sweep netting to collect flying insects, or those hiding in the foliage, was made using a sweep net (37 cm diameter) along a permanently flagged 30 m transect route in each plot once per month between June and August in both years. The surveyor walked at a constant speed, sweeping the net in an arc covering a 150° to 180° side to side through the vegetation, covering an area of approximately 1 m on either side of the transect. The surveyor always attempted to carry out sweeping into the sun so that shadows were cast behind and did not cause flying insects to flee.

Pan traps were deployed during each visual survey and sweep netting sampling event. Following recommendations by Vrdoljak & Samways (2012), these traps consisted of white and yellow colored bowls (18 cm diameter), which are particularly attractive to many species of Diptera and Hymenoptera. Each bowl was filled approximately halfway with water mixed with a small amount of unscented dish detergent. Ten bowls were place on the ground approximately 3 m apart, alternating in colour, along a transect and left out for approximately 30 min while the visual or net sampling occurred.

Visual surveys, sweep netting, and pan trapping only occurred during ideal weather conditions (i.e., low wind, warm temperature > 15 °C, and no precipitation). All specimens collected through net sweeps and pan traps were stored in alcohol for temporary preservation and were later pinned, spread (as required), dried, and stored at the University of Guelph Insect Collection (Guelph, ON, Canada). All pollinators observed were documented including Lepidoptera (butterflies and moths), Hymenoptera (bees, wasps, sawflies, and ants), Coleoptera (beetles), and Diptera (flies). Although the focus of the surveys was on quantifying butterfly and bee richness and abundance, all arthropods were recorded. For butterflies, bees, and other easily identifiable specimens, species or genus names were recorded when possible. Observations of other species such as Hymenoptera, Coleoptera, and Diptera were only identified to broader taxonomic groups such as Order or Family.

Bees were identified to genus using Packer et al. (2007) and to species using Mitchell (1960, 1962), and Sheffield et al. (2011). Wasps were identified using Goulet & Huber (1993), Buck et al. (2008), and Bohart & Menke (1976). Flies were identified using Marshall (2012), McAlpine et al. (1981, 1987), Miranda et al. (2013), and Thomas (2011). Beetles were identified to family-level using Marshall (2018). Other insect families were identified using Marshall (2017), Kelton (1980), and Vickery (1985). Insects were placed into feeding guilds using Marshall (2017).

Canada Overview

Yukon

N.W.T.

Nunavut

British
Columbia

Alberta   Sask.   Manitoba

N.L.

Ontario

Quebec

N.B.

USA

Mexico

Alberta

Site 4

Site 5

Site 3

Site 1

Site 2

Donnelly

0   20   40   60   80   100
Kilometers

Saskatchewan

Site 7

Site 11
Site 10

Site 9

Site 8

Moose Jaw

Regina

0   20   40   60   80   100
Kilometers

Manitoba

Site 17

Site 16

Site 14

Site 15

Site 13

Brandon

0   20   40   60   80   100
Kilometers

Study site locations within each province. Stars represent the location of each study site within their respective province and square points represent the location of each pair of treatment and control plots

Statistical analysis

Similar to Campbell et al. (2019), to create a complete sample of the overall community through time, all monthly observations from the three different sampling techniques were pooled together into a single yearly count per species per plot. We also emulated Campbell et al. (2019) by classifying insects by feeding guild (pollinator, predator, or herbivore). Due to the lack of control plots in 2018, we included only 2019 data ($n = 30$) to analyze the effect of treatment on guild abundance (number of individuals) and family richness (number of unique taxonomic families). This same approach was also used to assess the effect of plantings on wild bee abundance and richness. We used likelihood ratio tests (Chi-square) of full and reduced generalized linear models to test for differences between plot treatment in guild abundance and richness. Predicator variables included within each full model were plot treatment (planted vs control), province (AB, MB, SK) and plot size (area in ha). Because pollinators were of particular interest and to determine which species might be driving trends in the pollinator guild overall, we repeated the abundance analysis for each pollinator family that contained ≥ 1% of the total pollinator abundance in 2019 (Morandin & Kremen 2013; see Table 1).

Finally, we investigated changes in guild and wild bee abundance, and pollinator richness, through time for the subset of plots with data collected in both years ($n = 18$). Generalized linear mixed effect models were used to assess both abundance and richness, and fixed effects included year (2018, 2019), province (AB, SK, MB), and plot size. We created models that included every combination of these fixed effects. In addition, plot ID was included in each model as a random effect to account for multiple observations per plot. We then used Akaike Information Criteria model selection, corrected for sample size (AICc), to determine the most parsimonious model. Change between years in predator, herbivore, and wild bee richness were visually assessed, since small sample sizes in these groups made statistical models unfeasible.

For all analyses of abundance, we used negative binomial family models with a log-link function to account for overdispersion in the data. Richness data did not display overdispersion and so Poisson family models with a log-link function were used. All analyses were conducted in R v4.0.2 (R Core Team 2020). Negative binomial family models were calculated using the R package "MASS" (Venables & Ripley 2002), mixed effect models using the R package "lme4" (Bates et al. 2015), model diagnostics were assessed using the R package "DHARMa" (Hartig 2020), and AICc model comparison completed using "bbmle" (Bolker & R Development Team 2021). Results are reported as mean ± SE unless otherwise noted.

Results

Overall, arthropod captures totalled 3,883 in 2018 and 5,792 in 2019. Combined over both years, we captured arthropods from 115 different taxonomic families. The most common families were Apidae (18% of total captures; bees), Syrphidae (8% of total captures; syrphid flies), Pieridae (7% of total captures; butterflies), Anthomyiidae (4% of total captures; root maggot flies), and Halictidae (4% of total captures; sweat bees). In addition, unidentified Diptera (i.e., flies not identified beyond order) made up 8% of total captures. Predator captures were predominantly dragonflies and damselflies (Odonata: 51% and 28% of all predators, respectively) with only a few spiders (Aranea: $n = 8$) observed, while herbivores consisted of grasshoppers and crickets (Orthoptera: 53%), plant bugs (Miridae: 17%), and leafhoppers (Cicadellidae: 10% of all herbivore captures). Grouped by feeding guild, captures consisted of 72% pollinators, 8% predators, and 20% herbivore arthropods in 2018 and 86%, 12%, and 2% respectively in 2019. The most commonly observed bees were Bombus (65% of all bee captures) followed by sweat bees (4% Lasioglossum + 2% Halictus + 10% other or unidentified to genus Halictidae). Very few honey bees (*Apis mellifera*) were observed (40 individuals in 2018 and 65 in 2019) and these observations were excluded from all analyses of wild bees.

Pollinator abundance was significantly higher in planted plots than control plots in 2019 (Fig. 2A). Abundance was significantly related to plot treatment ($X^2 = 5.38$, df = 1, $p = 0.02$) and province ($X^2 = 15.1$, df = 2, $p < 0.001$) but not plot size ($X^2 = 0.47$, df = 1, $p = 0.49$). On average, planted plots had 209 ± 42 pollinators compared to 124 ± 18 in control plots. Pollinator abundance significantly differed between Manitoba and Alberta ($\beta \pm SE = -0.84 \pm 0.24$, z = -3.5, $p < 0.001$) but not Alberta and Saskatchewan ($\beta \pm SE = 0.20 \pm 0.24$, z = 0.8, $p = 0.41$). Manitoba had lower mean pollinator abundance (80 ± 15) than both Alberta (185 ± 36) and Saskatchewan (235 ± 50). Planted plots had significantly higher abundance of the pollinator families Andrenidae (mining bees), Apidae (honey, bumble, long-horned, orchid, and digger bees), and Halictidae (sweat bees), and significantly lower abundance of Cleridae (checkered beetles; predominantly *Trichodes*, *Trichode nutalli*) compared to control plots (Table 1).

Wild bee abundance and richness was higher in planted plots (Fig. 2). Abundance was significantly related to plot treatment ($X^2 = 7.85$, df = 1, $p = 0.005$) and province

Table 1 Mean (SE) abundance of pollinator family/group captured in flower-planted and control plots in 2019. Analysis only included pollinator families or groups whose abundance was $\geq 1\%$ of total 2019 pollinator captures. Bold means (SE) in either control or planted plots are significantly different than the means in the other plot type. Treatment effect p-values (P_{TRT}) are included

| Order	Family/group	P_{TRT}	Total captures	Control	Planted
Coleoptera	Cleridae	**0.012**	36	**2.0 (0.8)**	**0.4 (0.1)**
Diptera	Anthomyiidae	**0.009**	120	**2.3 (0.8)**	**5.7 (1.9)**
	Bombyliidae	0.72	34	1.1 (0.6)	1.1 (0.4)
	Muscidae	0.08	100	4.3 (1.9)	2.4 (1.4)
	Sarcophagidae	0.25	169	4.5 (1.1)	6.8 (1.8)
	Syrphidae	0.52	597	17.9 (6.0)	21.9 (7.1)
	Tachinidae	0.80	59	1.9 (0.6)	2.1 (0.6)
	Unidentified Diptera	0.81	694	20.9 (5.7)	25.4 (14.3)
Hymenoptera	Andrenidae	**0.004**	78	**1.2 (0.4)**	**4.0 (1.2)**
	Apidae	**0.014**	1391	**14.9 (2.8)**	**77.9 (33.6)**
	Colletidae	0.74	50	1.4 (0.8)	1.9 (0.7)
	Crabronidae	0.71	78	2.3 (1.0)	2.9 (1.0)
	Halictidae	**0.03**	170	**3.3 (0.9)**	**8.0 (2.3)**
	Megachilidae	0.44	87	2.7 (1.0)	3.1 (0.7)
	Sphecidae	0.96	40	1.0 (0.4)	1.7 (1.1)
	Vespidae	0.08	93	4.6 (2.2)	1.6 (0.5)
Lepidoptera	Crambidae	0.99	164	5.5 (1.0)	5.5 (1.5)
	Hesperiidae	0.90	73	2.4 (0.7)	2.5 (1.0)
	Lycaenidae	0.95	164	5.6 (1.7)	5.3 (1.1)
	Nymphalidae	0.59	171	5.1 (1.4)	6.3 (1.6)
	Pieridae	0.50	406	12.9 (3.2)	14.1 (2.5)
Neuroptera	Chrysopidae	0.16	31	0.5 (0.4)	1.6 (1.0)

($X^2 = 19.8$, df $= 2$, $p < 0.001$) but not plot size ($X^2 = 0.02$, df $= 1$, $p = 0.88$). On average, planted plots had 93 ± 34 bees compared to 22 ± 4 in control plots. Bee abundance significantly differed between Saskatchewan and Alberta ($\beta \pm SE = 1.31 \pm 0.35$, $z = 3.8$, $p < 0.001$) but not Alberta and Manitoba ($\beta \pm SE = -0.50 \pm 0.35$, $z = -1.4$, $p = 0.15$). Saskatchewan wild bee abundance averaged 129 ± 48, while Alberta was 26 ± 4, and Manitoba 16 ± 4.

Predator abundance was also significantly higher in planted plots in 2019 (Fig. 2A). Abundance was significantly related to plot size ($X^2 = 5.72$, df $= 1$, $p = 0.02$) but not plot treatment ($X^2 = 3.58$, df $= 1$, $p = 0.058$) or province ($X^2 = 3.50$, df $= 2$, $p = 0.17$). On average, each unit increase in plot size resulting in predator abundance decreasing by 0.40. Herbivore abundance in 2019 did not differ by plot treatment ($X^2 = 0.08$, df $= 1$, $p = 0.78$), province ($X^2 = 4.59$, df $= 2$, $p = 0.10$), or plot size ($X^2 = 1.39$, df $= 1$, $p = 0.24$).

While there was no effect of treatment, province, or plot size on richness for any guild (results not shown), the richness of wild bee genera followed a similar pattern to bee abundance, where richness was significantly related to plot treatment ($X^2 = 4.42$, df $= 1$, $p = 0.035$) and province ($X^2 = 11.7$, df $= 2$, $p = 0.003$) but not plot size ($X^2 = 1.02$, df $= 1$, $p = 0.31$). On average, planted plots had 7 ± 1 genera compared to 5 ± 1 in control plots (Fig. 2B). Wild bee richness significantly differed between Manitoba and Alberta ($\beta \pm SE = -0.44 \pm 0.20$, $z = -2.2$, $p = 0.028$) but not Alberta

and Saskatchewan ($\beta \pm SE = 0.21 \pm 0.17$, $z = 1.2$, $p = 0.24$). Manitoba had lower mean bee richness (4 ± 1) than both Alberta (6 ± 1) and Saskatchewan (7 ± 1).

Overall, pollinator and predator abundance were similar between years while herbivore abundance declined significantly (Fig. 3A). The most parsimonious model (i.e., ΔAICc < 2) of pollinator abundance included only the fixed effect of Province (Table S2). In this model, the only significant predictor was province (MB: $p < 0.001$, reference province: AB) and there was no effect of year. Similarly, year had no effect on predator abundance where the most parsimonious models were the null model containing only the random effect of plot and the models with the fixed effect of Year and Province (Table S3). The most parsimonious models for herbivore abundance included the fixed effects Year and Year + Province (Table S4). Significant predictors in these models were year ($p < 0.0001$) and province (SK: $p < 0.05$). Herbivore abundance declined from an average of 44 ± 4 in 2018 to 2 ± 0.5 in 2019 (Fig. 3A). Finally, the most parsimonious models for wild bee abundance were the models with the fixed effects of Year + Province, Year + Province + Area, and Province (Table S5). Year was a significant predictor (2019: $\beta \pm SE = 0.65 \pm 0.27$, $z = 2.4$, $p = 0.018$) while province (MB: $\beta \pm SE = -0.81 \pm 0.42$, $z = -1.9$, $p = 0.06$; SK: $\beta \pm SE = 0.82 \pm 0.43$, $z = 1.9$, $p = 0.6$) and area ($\beta \pm SE = 0.59 \pm 0.42$, $z = 1.4$, $p = 0.15$) were not.
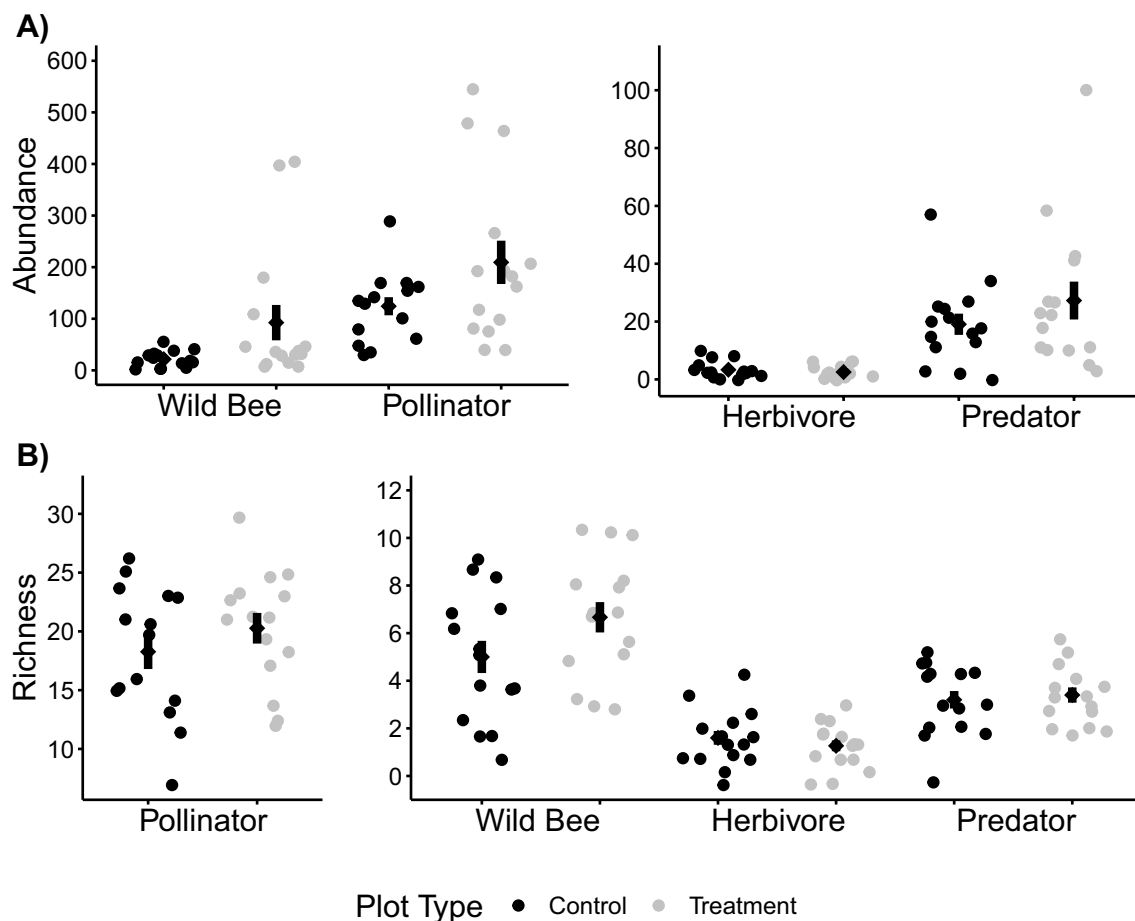
Fig. 2 Total abundance (**A**) and richness (**B**) per plot in 2019 for each group by treatment. Abundance is number of individuals while richness is number of genera for wild bees and otherwise the number of taxonomic families for insect guilds. Black diamond points represent the mean and error bars represent SE. Points are jittered on the x-axis to prevent overlap and improve readability

Bee abundance increased from 41 ± 9 in 2018 to 79 ± 29 in 2019 (Fig. 3A).

In general, all three guilds' richness declined or was stable between years while wild bee richness increased (Fig. 3B). The most parsimonious models of pollinator richness included the fixed effects Year + Province and Year + Province + Area (Table S6). Year ($p < 0.005$) and province (MB: $p < 0.05$) were significant but plot size was not. Pollinator richness decreased from an average of 23 ± 2 families in 2018 to 19 ± 1 in 2019.

Discussion

Planted flower plots supported a greater abundance of pollinators, especially wild bees, than out-of-production agricultural areas. These results are consistent with published research from agricultural landscapes in other parts of North America and in Europe. For example, Campbell et al. (2019) found higher taxa richness and higher abundance of pollinators, predators, and herbivores in planted wildflower plots than in fallow agricultural plots within Florida, USA. Williams et al. (2015) observed at least 11% higher species richness and over 60% higher wild bee abundance in wildflower plots than unmanaged control areas in Florida, Michigan, and California, USA. Similarly, Grass et al. (2016) reported wildflower plantings supported highly diverse pollinator communities within central Germany agricultural landscapes. Our findings support the use of flower plantings as an effective approach to conserve and promote insect pollinator communities within agricultural landscapes.

The size of research plots, which ranged from 0.13 to 1.70 ha (0.3–4.2 acres), did not have a significant effect on abundance or richness, suggesting that even small-scale flower plantings can have a positive effect on pollinator communities. Small-scale restoration of hedgerows has been shown to provide conservation benefits for sensitive and rare pollinator species within intensive agricultural landscapes (Kremen & M'Gonigle 2015). In addition, local wildflower enhancements have been shown to significantly improve the

Fig. 3 Change in abundance (**A**) and richness (**B**) between years per group. Abundance is number of individuals while richness is number of genera for wild bees and otherwise the number of taxonomic families for insect guilds. Points represent each plot's abundance or richness in each year, with lines connecting each plot's points to help visualize the direction of change between years per plot. Point shape and line color represents province (AB = Alberta, MB = Manitoba, SK = Saskatchewan). Points are jittered on the x-axis to improve readability
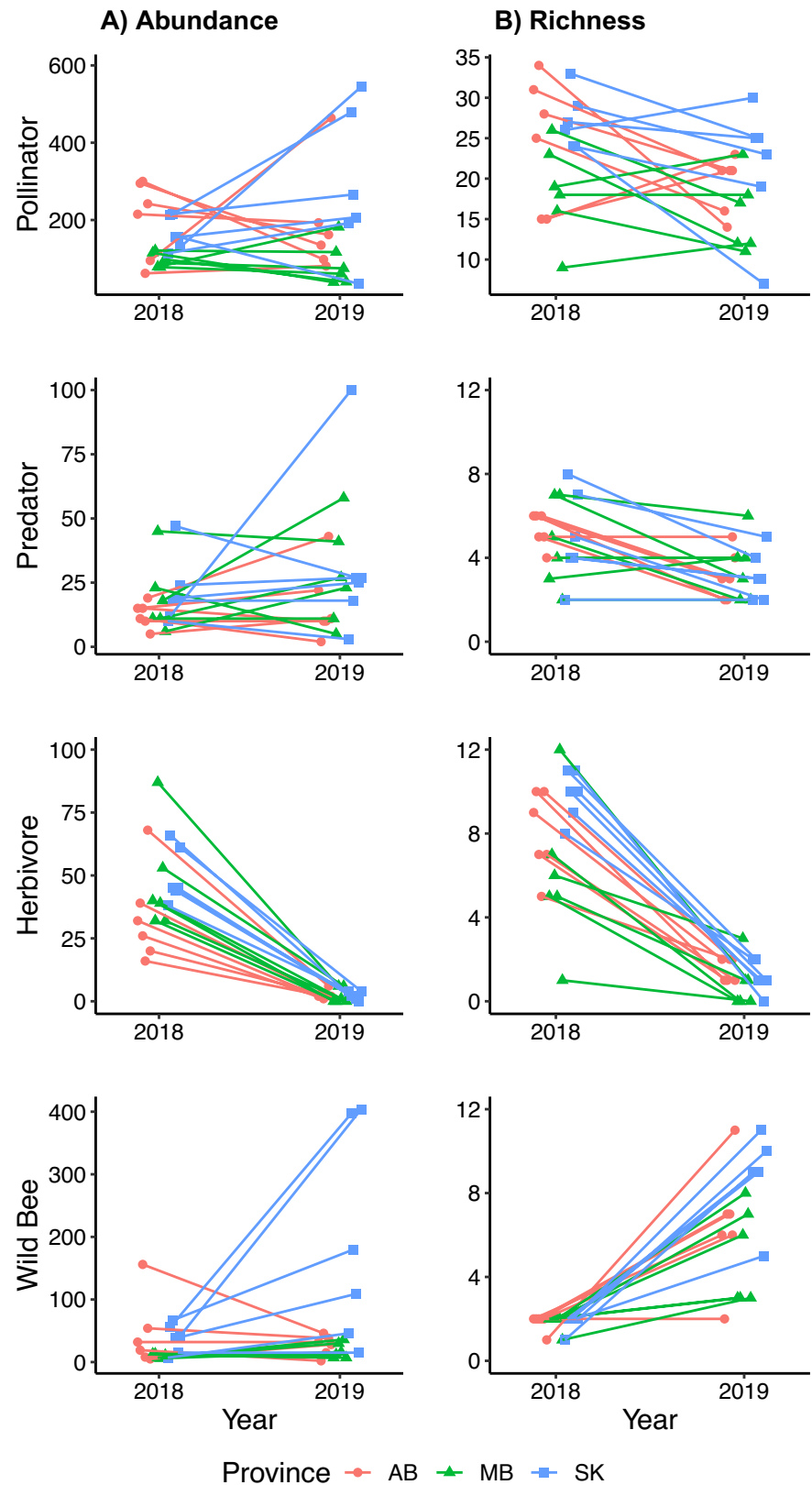
surrounding landscape-scale abundance of wild bees (Kleijn et al. 2018). The effectiveness of small-scale flower plantings to increase pollinator abundance and richness may alleviate potential concerns regarding the need to commit large parcels of land to conservation projects and may encourage increased participation from private landowners.

Changes in arthropod abundance and richness between years were likely driven by environmental conditions. Abundance and richness of predators was stable between years while herbivores declined steeply. Similarly, pollinator abundance was stable, though richness declined. Interestingly, the subset of pollinators consisting of wild bees increased in both abundance and richness suggesting the suitability of flower habitat varied by plant species. Weather conditions were also a likely factor in low arthropod abundance and declines in richness in 2019. In general, weather across the study sites in 2019 was very dry throughout the spring followed by below average temperatures and excessive precipitation in late summer and fall (Bezte 2020; Environment Canada 2020). While some declines in arthropod abundance were evident in our study between years, the increase in wild bee abundance suggests that the planted plots worked to buffer the effects of inclement environmental conditions on at least some pollinator species within the community, allowing them to persist and grow under adverse settings. Unfortunately, additional surveys planned for 2020 and 2021 were cancelled due to health and safety concerns related to the COVID-19 global pandemic, preventing further investigation of trends over a longer time span.

Pollinator community response to floral plantings has been shown to be dependent on the floral species selected for planting (Williams et al. 2015) and the seed mix used in this pilot project favored plants that are beneficial to wild bees. Including a larger number or variety of flowering species in restoration plantings may improve outcomes for a wider range of insect pollination species, including the need for plants that support less common pollinator species (Williams et al. 2015; Havens & Vitt 2016). It seems likely the seed mix used in the pilot program could be significantly improved by increasing the diversity of plant species planted, though this could result in an increased cost of the seed mix. Finding the balance between creating optimal pollinator habitat restoration with planting cost-effectiveness is key to effective land restoration management (Wilkerson et al. 2014; Harmon-Threatt & Hendrix 2015; Meissen et al. 2019).

In some cases, arthropod abundance and richness differed between provinces, though reasons for this are unclear. Differences between provinces could have been driven by any number of factors including variation in environmental or surrounding habitat conditions, differences in existing local arthropod populations affecting colonization rates, or differences in surrounding agricultural practices. Another factor to consider is that, in addition to food resources, pollinators need habitat for breeding and winter hibernacula, without which they are unlikely to persist or thrive within a given area (Ellis & Barbercheck 2015; Ganser et al. 2018; Requier & Leonhardt 2020). Unploughed flower plots can provide some species with nesting or overwinter habitat (Ganser et al. 2019) but appropriate habitat features vary by species (e.g., ground versus woody debris nesting bees). A conservation approach beyond just improving floral resources would be required to enhance pollinator communities over the long term if areas lack sufficient nesting or overwinter habitat (Requier & Leonhardt 2020).

The flower species that successfully promote pollinator communities have been shown to vary according to region (Williams et al. 2015; Campbell et al. 2019; Liczner & Colla 2020). It is possible that the flower species selected for planting were better adapted to some local conditions than others, but we were unable to quantify differences in vegetation growth and floral resources between sites due to the lack of vegetation surveys conducted during the pilot project. Plant species provenance can also have a significant effect on pollinator-flower interactions (Bucharova et al. 2021). This highlights uncertainty about the effectiveness of applying a single conservation approach across a broad geographic region. Ideally, pollinator conservation plantings would use native plant assemblages matched to local conditions to achieve habitat restoration goals (Havens & Vitt 2016; Meissen et al. 2019).

A common and important question is whether flower plantings export insect pollinators to neighbouring areas or concentrate them within the preferred habitat to the detriment of the surrounding landscape (Kleijn et al. 2011; Kremen & M'Gonigle 2015). If flower plots were exporting pollinators to nearby areas, then we would expect to find greater pollinator abundance in control sites that are closer to planted plots than control sites that are further away. Preliminary analysis of our data did not show this relationship, but the small sample size and short time frame of the study was not well suited to answer this question. Similar conservation projects have found evidence that restoration of pollinator habitat does lead to exportation of pollinators to surrounding areas. For example, both Blaauw & Issacs (2014) and Ganser et al. (2018) report that wildflower strips planted alongside fields of crops resulted in enhance crop pollination, indicating that pollinators were being exported from restored areas. Similarly, Kremen and M'Gonigle (2015) report that restoring hedgerow habitat within intensive agricultural areas increased the number of flower visitors in the fields immediately adjacent to those hedgerows. Further research over a longer time frame is required to properly assess whether concentration of pollinators is occurring within the plots in our study.

Operation Pollinator is intended to enhance pollinator habitat, promote pollinator populations, and increase biodiversity within agricultural landscapes. Our results indicate that the initiative to plant floral resources within Canadian agricultural landscapes supports these program goals. Flower-planted plots supported greater pollinator and especially wild bee abundance, and higher wild bee richness than other areas. These positive effects on local pollinator communities were achieved at small scales using a relatively simple floral seed mix (six species), which likely could be further improved by increasing the diversity of plant species included or matching the restoration plantings to local environmental conditions and native plant communities. Regardless, we find that flower plantings have a clear benefit to local pollinators communities and are an effective conservation method to implement within agricultural landscapes.

Author contributions Study design was conducted by JEL and DRN. Data collection facilitated by GK. Data analysis conducted by SGV with input from DRN. The first draft of the manuscript was written by SGV and all authors reviewed previous versions of the manuscript.

Declarations

Conflict of interest The authors declare no conflict of interest.

Ethical approval Not applicable.

Consent to participate Not applicable.

Consent for publication Not applicable.

Data availability The data analysed for this study is available from the corresponding author on reasonable request.

Code availability Not applicable.

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