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Pollinator response to prairie strips in the Iowa agricultural landscape

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Pollinator response to prairie strips in the Iowa agricultural landscape

by

Caroline J. Murray

A thesis submitted to the graduate faculty
in partial fulfillment of the requirements for the degree of

MASTER OF SCIENCE

Co-majors: Sustainable Agriculture; Entomology

Program of Study Committee:
John C. Tyndall, Co-major Professor
Matthew E. O'Neal, Co-major Professor
Peter T. Wolter

The student author, whose presentation of the scholarship herein was approved by the program of study committee, is solely responsible for the content of this thesis. The Graduate College will ensure this thesis is globally accessible and will not permit alterations after a degree is conferred.

Iowa State University

Ames, Iowa

2021

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ABSTRACT

In recent decades, several families of pollinators have experienced significant population losses in North America, including the Monarch butterfly, honey bee, and a several native bee species. Pollination via honey bees alone is worth over \$15 billion per year attracting considerable attention to developing solutions to reverse the current trend. Potential causes for this decline are habitat loss and fragmentation, insecticide use, and pests and diseases. Declining pollinator communities in the Midwest have triggered research on effective conservation methods that can be integrated into an agricultural landscape dominated by corn and soybean row crops. There are many conservation practices with cost-share opportunities for farmers available through the Farm Bill, though many focus primarily on improving water quality while only one was contrived with pollinators as the priority. Our first study was designed to investigate the impacts of one water-quality centered practice, prairie strips, on communities of monarchs, native bees, honey bees, and syrphids in central Iowa. The results of this study could be used to inform policy-makers, conservationists, and agricultural producers alike, providing valuable insight into the multi-faceted effects of prairie strips. Understanding these impacts could assist in targeted conservation, specifically where ecosystem services (i.e. pollination for crop production or apiary location) may be of need. In a second study, we developed spatial analytical methods to explore potential landscape-level effects of land use on abundance and diversity within insect pollinator communities. In addition, we analyzed Conservation Reserve Program's CP-42 patch acreage and connectivity on the state and county level to better understand the organization of this land cover on the Iowa landscape. The overall aim of this project was to broaden our knowledge of the impacts of native vegetation, specifically prairie

strips, on a variety of pollinator guilds, thereby informing future pollinator conservation efforts in the Midwest

CHAPTER 1: GENERAL INTRODUCTION

Across North America, certain pollinators are showing signs of decline with habitat loss, insecticide use, and pests/diseases cited as likely causes (Goulson et al. 2015). This loss of pollinators has affected both native and non-native taxa. For non-native honeybees, Varroa mite (Steinhauer et al. 2018) and colony collapse disorder have plagued the species across the U.S., in particular the Midwest causing greatly reduced colony survival (Kulhanek et al. 2017). Monarch butterflies, a native taxa, are also experiencing population loss, though potentially due to the decline of their obligate host, milkweed (*Asclepias spp.*) (Pleasants et al. 2017, Pleasants & Oberhauser 2013, Flockhart et al. 2015), degradation of their overwintering grounds (Vidal et al. 2013), and limited nectar resources along their migration route (Brower et al. 2006, Inamine et al. 2016). Finally, certain native bee species, including members of the genus *Bombus*, are declining in the Midwest as well (Grixti et al. 2009). Habitat loss is one driver that is cited for the decline of all of these taxa, and it is happening across the Midwest, especially as agricultural production shifts toward a near-monoculture (Otto et al. 2016, Plourde et al. 2013).

The potential for losing these pollinator taxa is an enormous matter. While the most prominent crops in Iowa (i.e. corn and soy) do not require insect pollination, many economically important crops grown for food production do need these pollination services (Reilly et al. 2020). As pollinator decline looms ominously over crop production in North America, research has ramped up to find effective conservation solutions. Conservation efforts are ongoing in the form of roadside plantings, urban pollinator gardens, and others including CP-42 through the USDA conservation reserve program. Many conservation practices target native, perennial vegetation as a key factor in improving pollinator habitat (Asbjornsen et al. 2014).

Prairie strips are one practice that could reintroduce native perennial vegetation to the homogenized Midwest agricultural landscape. Studies have found that prairie strips improve water quality (Gutierrez-Lopez 2014), increase bird biodiversity (Schulte et al. 2017), and support native bee communities (Kordbach et al. 2020). There has not yet been any examination of the impact of prairie strips on monarch butterfly abundance or the non-Hymenoptera pollinator community (i.e. syrphids) particularly in the presence of apiaries. Studies have shown that managed honey bees respond positively to CRP plantings. (McMinn-Sauder et al. 2020, Ricigliano et al. 2019). We conducted this study to determine to what extent certain pollinators respond to the high-diversity native perennial vegetation resources available in prairie strips.

The goal of this thesis was to complete field studies to determine the impact of prairie strips on a variety of pollinator taxa. In Chapter 2, we report the results of the pollinator field research noting novel observations and potential applications to pollinator conservation. In Chapter 3, we design and test spatial analytical methodology to study pollinator habitat on a larger spatial scale, both to assess potential confounding variables that may have impacted our field research and also to understand the spatial connectivity of conservation practices on the Iowa landscape. Chapter 5 provides a comprehensive conclusion of the research and results presented in this thesis and suggestions for further research.

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CHAPTER 2: PRAIRIE STRIPS INCREASE ABUNDANCE OF FLORAL RESOURCES AND KEY POLLINATORS.

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Abstract

The recent decline of several pollinator taxa has motivated conservation efforts within the agricultural realm. Integrating patches of native tallgrass prairie into corn and soybean fields using prairie strips is one practice that may reverse pollinator habitat loss. We hypothesized that applying this practice to corn and soybean fields with prairie strips will increase floral resources, native bees (Hymenoptera: Apoidea), syrphids (Diptera: Syrphidae), and monarch butterflies (Lepidoptera: Nymphalidae, *Danaus plexippus*) than fields without. Because apiaries were present at all field sites, we hypothesized that there would be no difference in the observed population of honey bees (Hymenoptera: Apidae, *Apis mellifera*). In our study, we sampled native bees, syrphids, and honey bees assessing activity-density and richness at both site types. We also surveyed floral resources, including milkweed (*Asclepias spp.*) and adult monarchs residing around them. We found over twice the richness and abundance of floral resources at prairie strips than control sites, suggesting enhanced pollinator forage. Adult monarchs were observed in greater numbers at prairie strips as well, with activity-density increasing significantly from June to August, despite finding more milkweed at control sites. Native bee

activity-density and richness was higher at prairie strips in June, but higher in control sites during July, August, and September. Like native bees, syrphid activity-density was also higher at prairie strips in June. Syrphid richness did not vary between site types. Honey bee activity-density was higher at prairie strips sites in July, with an inverse trend in August and September. These results suggest that while prairie strips may not provide the milkweed necessary for monarch reproduction, they do provide critical forage for certain pollinators at different periods during the growing season, notably monarchs during their late season migration.

Key words

Conservation, syrphids, prairie strips, monarchs

Introduction

Pollinators provide an exceptionally important ecosystem service and are currently imperiled worldwide (Gilbert 2016). Insect pollinators, in particular, native bees [Hymenoptera: Apoidea] and honey bees [Hymenoptera: Apidae, *Apis mellifera*] all pollinate economically important crops with many high value crops showing signs of being pollination limited (Reilly et al. 2020), risking both farm economies as well as food security (Winfrey et al. 2008). Not only are certain cropping systems at risk, considering that 87% of all flowering plants are insect pollinated, the implications to other ecosystem services are potentially extensive and deeply concerning (Christmann 2019).

Pollinator decline is apparent in the in the US Midwest, particularly Iowa, where agriculture has become increasingly industrialized over the last several decades (Plourde et al. 2013). In an attempt to maximize crop yields, farmers have increased their use of fungicides, herbicides, and insecticides, particularly neonicotinoids, which are under scrutiny for their

potentially negative impact on bees and other insect pollinators (Goulson et al. 2015, Henry et al. 2012). Agricultural intensification minimizes the availability of blooming flowers and specific pollinator host species (e.g., *Asclepias spp.*) to monarch butterflies [Lepidoptera: Nymphalidae, *Danaus plexippus*], native bees, honey bees, and syrphids [Diptera: Syrphidae], likely contributing to population declines of these pollinators (Semmens et al. 2016, Pleasants and Oberhauser 2013).

Reversing the decline of pollinators drives both policy and research to evaluate the effectiveness of enhancing habitat to increase the availability and accessibility of pollen and nectar (Wratten et al. 2012), and concomitant best practices that minimize habitat degradation (Ehmke et al. 2015). Conservation practices that increase perennial vegetation with the goal of enhancing water quality or protecting soil, also have the capacity to create and enhance pollinator habitat depending on how those practices are established and managed (Wratten et al. 2012, Burkle et al. 2017). In this study, we evaluated how a new water quality oriented agricultural conservation practice, prairie strips, impacts the abundance and richness of several insect pollinator families.

One way to reintroduce pollinator habitat into crop fields is through the integration of highly diverse native perennial vegetation. To accomplish this, many landowners utilize the conservation reserve program (CRP), which provides cost-share for those incorporating native vegetation into farm fields. Many CRP practices boast environmental benefits including better water quality, enhanced plant biodiversity, and positive impacts on honey bee colony growth (Ricigliano et al. 2019, McMinn-Sauder et al. 2020). Based on extensive long-term research led by Iowa State University, the Agriculture Improvement Act of 2018 (colloquially known as the 2018 Farm Bill) authorized CP-43 or “Prairie Strips” as a conservation practice eligible for CRP

funding. Prairie strips are linear arrangements of native prairie grasses and forbs planted in the form of in-field contour buffer strips and edge-of-field filter strips that intercept run-off from the adjacent crop field (USDA FSA 2019). Prairie strips reduce nutrient and sediment loss (Gutierrez-Lopez 2014), increase bird biodiversity (Schulte et al. 2017), and improve honey bee colony weight (Zhang 2020). In addition, Kordbacheh et al. (2020) found that prairie strips harbor a diverse native bee community, supporting a higher abundance of native bees than either corn or soybean fields. Although corn and soybean fields can harbor several species of pollinators (Wheelock et al. 2016), these annual crops likely provide limited resources throughout the growing season. The extent to which prairie strips supports pollinators beyond the resources found in existing non-cropped features of the agricultural landscape has not been documented. Iowa consists of ~64% corn and soy fields, with a limited amount of more perennial, non-cropped habitat like roadsides, fence-lines or grass waterways (NASS-USDA 2017).

Agricultural lands are also a common site for honey beekeeping, and honey bees utilize the native plants found within prairies (Carr-Markell et al. 2020, Tuell et al. 2008, Zhang et al. 2021). The extent to which non-native, managed honey bees may capitalize on the small patches of native prairie that are established via CP-43 practices is not yet known. When honey bees share resources with native bees, there is the potential for negative interactions (Pritchard et al. 2021). Kordbacheh et al. (2020) did not consider the potential impacts of managed honey bee colonies on native bee communities in prairie strips as they selected locations where honey bees were not being kept. There is increasing interest in determining the impact of apiculture on efforts to conserve native pollinators (Mallinger et al. 2017). Although honey bees have been found in both corn and soybean (Wheelock et al. 2016), their presence is not guaranteed unless

managed colonies are close to these fields (St. Clair et al. 2020). By including honey bee apiaries at our field sites, we can better understand honey bee usage of prairie strips.

Considering that prairie strips are planted with a high diversity seed mix, we hypothesized that prairie strips would have more pollinator habitat in the form of milkweed and blooming flowers than control sites. Based on the observed impacts of prairie strips on other pollinating insects, we hypothesized that there would be more monarchs, native bees, and syrphids at farm fields with prairie strips than fields without. We also hypothesized that farm fields with prairie strips would have more native bee and syrphid species richness. To test these hypotheses, we collected native bee, honey bee, and syrphid specimens using bee bowls. We did Pollard walks to measure adult monarch abundance and strip transect surveys to quantify milkweed and blooming flowers. We predicted that integrating perennial native vegetation (i.e. prairie strips) into farm fields there would be an increase in the abundance and richness of the pollinators we studied.

Materials and Methods

Site selection

Due to the field-scale implementation of prairie strips, we used individual, commercial farms committed to the conventional production of corn and soybeans (i.e. not certified organic) as our experimental unit. Fields with prairie strips (prairie strips sites) were selected based on a minimum establishment period of two years. Prairie strips were added to the farm bill in 2018; therefore, they were not widely adopted when this experiment began. We located four prairie strips sites in 2018, and an additional site in 2019 (5 total) (table S1). The number of prairie patches varied between prairie strip sites, so one was selected randomly at each location to

collect data (i.e. Pollard walks, strip transect surveys, and bee bowl sampling) (Zhang 2020). We assigned a control site for each prairie strips site by locating a corn or soybean field within the same county that was at least 3.2 km from the prairie strips site. This distance was chosen to control for potential honey bee foraging crossover between site types (Beekman and Ratniecks 2000, Carr-Markell et al. 2020, Zhang 2020). At control sites, Pollard walks, strip transect surveys, and bee bowl sampling were done in a grass waterway or road right-of-way directly adjacent to a field committed to the conventional production of corn and soybeans.

An apiary of four honey bee (*Apis mellifera*; Hymenoptera: Apidae) colonies were placed at both prairie strip and control sites. These colonies were created from “nucleus” colonies containing a queen, workers, brood and honey. All were Italian honey bees (*Apis mellifera ligustica*) purchased in Iowa. Colonies were assigned to apiaries based on weight, to control for variation in colony size between sites, with each colony comprised of approximately 7,000 adult honey bees. To respond to colony growth throughout the season, hive boxes were added as needed. Data on the productivity and survival of these colonies is provided elsewhere (Zhang 2020).

Plant Surveys

We counted blooming flowers at both site types using a 1x100 m strip transect. We tabulated all flowers and their associated species (forbs only), using a flower type designation (i.e. umbel, head, and ramet) due to extreme size differences in the inflorescences of perennial forbs. In these surveys, all milkweed (*Asclepias syriaca*, *Asclepias incarnata*, and *Asclepias tuberosa*) ramets were counted and grouped into blooming and non-blooming.

Monarch Surveys

Modified Pollard walks were done once a month from June until September, with approximately 30 days between surveys (Pollard and Yates 1993). We only did surveys on days considered favorable for monarch flight ($>70^{\circ}$ F, partly to full sun, no chance of rain), noting temperature, wind speed, and cloud cover during each observation. Each Pollard walk was conducted along a 100m long, 5m wide transect for 20-minutes. Observers counted all adult monarchs in the transect area taking care to not double count any individuals. In 2019, observers also recorded monarch actions at the time of observation (i.e. flying, nectaring).

Collecting Native Bees and Syrphids

We collected native bees from our sites using pan traps (“bee bowls”) placed along a transect that were activated every other week from June until September in 2018, and until early October in 2019 (Gill and O’Neal 2015, Wheelock and O’Neal 2016). The transects were located within the same prairie strip or road right-of-way/grass waterway where flower and monarch surveys were done. Three bee bowls of a white, blue and yellow color were placed on a t-post, with a total of four t-posts at each site. Multiple bowl colors were used because different insect genera are attracted to specific bowl colors, and we wanted to capture as much species richness as possible (Gill and O’Neal 2015). The first t-post with bee bowls was placed 30m from the apiary and were then placed 10m apart. The bee bowls were placed on t-posts at the height of the canopy of the surrounding vegetation (Wheelock and O’Neal 2016) and were activated for 24 hours at a time by filling the pan traps (Solo brand, 3.2oz) halfway with a water and soap solution (Dawn, Proctor & Gamble, Cincinnati, OH) (St. Clair et al. 2020). Bee bowls were only

activated on days when weather was considered favorable for bee flight (i.e no precipitation or very high winds) (Gill and O’Neal 2015).

Native Bee and Syrphid Identification

Native bees and syrphids were cleaned and dried using methods described in Droege et al. (2010) prior to being identified to species. We used dichotomous keys from DiscoverLife (Ascher and Pickering 2015) to identify native bees to the lowest taxonomic unit possible. Due to the difficulty of identifying *Lasioglossum* to species, the subgenus *Lasioglossum dialictus* was identified to morphospecies. To identify syrphids, we used the Key to Genera of Nearctic Syrphidae (Miranda et al. 2013) and “Field Guide to the Flower Flies of Northeastern North America” (Skevington et al. 2019). We retained a voucher collection of all syrphids and native bee specimens from this study are housed in the O’Neal Laboratory at Iowa State University Entomology Department.

Statistical Analysis

We used R studio (R Core Team 2019) and the “lme4” package (Bates et al. 2015) to perform generalized linear mixed effects analyses of the relationship between site type and abundance of monarch butterflies, abundance and richness of syrphids, native bees, and floral resources. For fixed effects, we used site type (prairie strips or control) and month into the model. Bee bowl data were binned by month due to the differences in sampling dates between 2018 and 2019. Because we only had observations from bee bowls on a single date in October 2019 (10/3/2019), we combined those data with September. Several of the pollinator types showed evidence of interactions between month and site type, so the model was adjusted to

include an interaction term for those groups. We added site to the model as a random effect. For pollinator types with month by site type interactions, least square means with Tukey adjustments were used to compare month-to-month abundance and richness differences using the “emmeans” package in R Studio (Lenth 2019). We created residual plots using this model to check for outliers from homoscedasticity or normality. To assess milkweed abundance, we used t-tests assuming unequal variance (Welch’s T-test) to compare average milkweed ramets per site type across the 2018 and 2019 field season.

Results

Plant Surveys

In total, we observed 13 species of flowering plants at control sites, and 36 species at prairie strips sites. At control sites, we counted 4,841 total blooming flowers, and 24,342 at prairie strips sites over the two-year study. *Medicago sativa* (alfalfa), *M. lupulina* (black medic), and *Chichorium* spp. (chickory) made up 75.7% of the total blooming forbs at control sites. *Symphyotrichum ericoides* (white heath aster), *Zizia aurea* (golden alexanders), and *Rudbeckia hirta* (black-eyed susan) made up 64% of the total flowers at prairie strips sites.

Richness of blooming flower species was significantly greater in prairie strips sites ($df = 68$, $z = 8.466$, $p < 0.001$) than control sites (Fig 1a). Species composition at control sites and strips sites was fairly disparate with only five species found at both site types. The effect of the prairie strip installation on blooming flower abundance was also significant ($df = 65$, $z = 39.255$, $p < 0.001$) (Fig 1b), with the greatest difference between control sites and prairie strips sites in September.

The average number of milkweed ramets was greater at control than prairie strips sites (24.74 ± 3.71 vs. 4.00 ± 1.34) ($t = 5.26$, $df = 34$, $p < 0.001$). This analysis included the combined number of *A. syriaca*, *A. tuberosa*, and *A. incarnata* ramets because they were the only milkweed species observed at the field sites.

Monarchs

The combined data from both years revealed that significantly ($df = 49$, $z = 4.281$, $p < 0.001$) more adult monarchs were observed at prairie strips than control sites. The presence of prairie strips increased the number of adult monarchs by an estimated 0.89 ± 0.21 individual monarchs per site per date. We saw the greatest difference between adult monarch observations in August (Fig 2), with the number of monarchs observed at prairie strips sites being significantly higher in August than in June ($z = 3.318$, $p = 0.003$). In 2019, we recorded the actions of the adult monarchs (i.e. flying or nectaring). Of the 69 observed adult monarchs, only 3 were nectaring; 2 at control sites in June, and 1 at prairie strips sites in August. These data include all monarchs observed during Pollard Walks, both within and outside the transect.

Bees

In total, we collected 2494 native bees from bee bowls over the course of the study with 1152 from prairie strips sites and 1342 from control sites. The number of specimens collected in bee bowls varies due to both the density of given species within an area, but also its activity at the point in time which can vary due to abiotic (i.e. temperature) and biotic factors (i.e. floral resources). In June, there was more ($z = -2.50$, $p = 0.0123$) native bee activity-density at prairie strips sites in June than control sites. This pattern was reversed in July, August, and September

(Fig 3a), when activity-density was higher at control sites ($z = 2.42, 2.98, 4.78, p = 0.016, 0.003, <0.001$). Post-hoc t-tests comparing mean activity-density of individual species revealed that a subset of native bee species varied in overall occurrence, with higher activity-density at control sites for *A. virescens* ($t = 2.17, df = 104.44; p = 0.03$) (Fig 3b) and *M. desponsa* ($t = 2.08, df = 68.995; p = 0.04$) (Fig 3e), while *H. ligatus* ($t = -1.99, df = 105.78; p = 0.05$) (Fig 3d) and *B. pensylvanicus* ($t = -1.99, df = 89.34, p = 0.05$) (Fig 3c) were higher at prairie strips sites.

We observed greater native bee richness at prairie than control sites in June ($z = -3.17, p = 0.0015$), and no such differences detected in July, August, or September (Fig 4). These individuals were comprised of 83 species from 21 genera (Table 1.1).

In addition to native bees, we also caught honey bees in the bee bowls. We observed significantly ($z = -3.251, p = 0.011$) higher activity-density of honey bees at control sites than prairie strips sites. At control sites, activity-density of honey bees trends upward throughout the field season, while at prairie strips sites, it remained at or below 2.5 ± 0.64 bees per bowl (Fig 5).

Syrphids

We observed a remarkable number of adult syrphids in bee bowls during 2018 and made a concerted effort to track their diversity and activity-density throughout 2019. In total, we observed 10 species (Table 1.2) which did not vary between site type (Fig 6b). Syrphid activity-density revealed two distinct generations in the 2019 season at both control and prairie strips sites based on the number of adults collected within the bee bowls (Fig 6a). Our generalized linear mixed effects model revealed an interaction between site type and month. A least squares mean comparison showed that more syrphids ($z = -7.69, p < 0.001$) were observed in prairie strips in June.

Toxomerus marginatus represented approximately 91% of syrphids captured in bee bowls. Due to their dominance in the dataset, we also analyzed activity-density and species richness without *T. marginatus* and found that activity-density was still higher at prairie strips in June ($z = -3.011$, $p = 0.0026$). Syrphid richness did not vary between site types.

Discussion

This study shows the potential for pollinator conservation within commercial farm fields using small patches of native perennial, flowering vegetation (i.e. prairie strips). Establishing tallgrass prairie patches is difficult, especially in an intensive agricultural landscape. Our results show that prairie strips had both higher richness and abundance of blooming flowers than the equivalent field-edge habitat (road rights-of-way and grass waterways), especially in the late season. These flowering resources could be a critical food source for insects like adult monarchs that are preparing for fall migration in the August and September (Brower et al. 2006).

In contrast to the notable difference in abundance reported by Kordbach et al. (2020), between prairie strips sites and controls, we did not observe an equivalent increase in native bee abundance and richness at prairie strips compared to our control sites. However, this is understandable because we compared prairie strips to habitat found in field-edges or grass waterways embedded within the crop matrix, while Kordbach et al. (2020) used sites within a crop field as their control. The difference in habitat types is starker when comparing perennial vegetation to annual crops. Furthermore, our study included four honey bee colonies at each research site which have the potential to impact native bee populations when the two share a habitat (Mallinger et al. 2017). When honey bees were kept in prairies, there were limited impacts observed in the community of native bees, though more viral infections were observed in bumble bees (Pritchard et al. 2021). We consistently found honey bees in bee bowls

throughout the growing season. To what extent their presence interacted with the native bee community is unclear. This interaction may not be best studied with bee bowls (Grundel et al. 2011, Cane et al. 2000), as the amount of bees collected by these traps may vary by species, as well as by the available flowering resources. For example, St. Clair et al. (2020) noted that the activity-density of honey bees estimated with bee bowls in soybean fields varied with the maturity of the crop. Bee bowls captured more honey bees as soybeans ceased flowering and senesced. To what extent the increasing amount of flowering resources in the prairie strips going into August and September reduced the relative attractiveness of bee bowls compared to those in control sites is unclear. Future exploration of how honey bees and native bees utilize prairie strips would benefit from direct observations of flowers.

The activity-density of syrphids was higher at prairie strips sites in June. This result could have a two-fold benefit to farmers who have prairie strips including both the benefit of increased pollination services, as well as more aphidophagous syrphid larvae, which are known to feed on the most economically damaging soybean pest in Iowa, *Aphis glycines* (Noma and Brewer 2008). Prairie strips have the potential to be an overwintering site for syrphids which could provide biological control for soybean pests in subsequent years.

Acknowledgments

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Figures and Tables

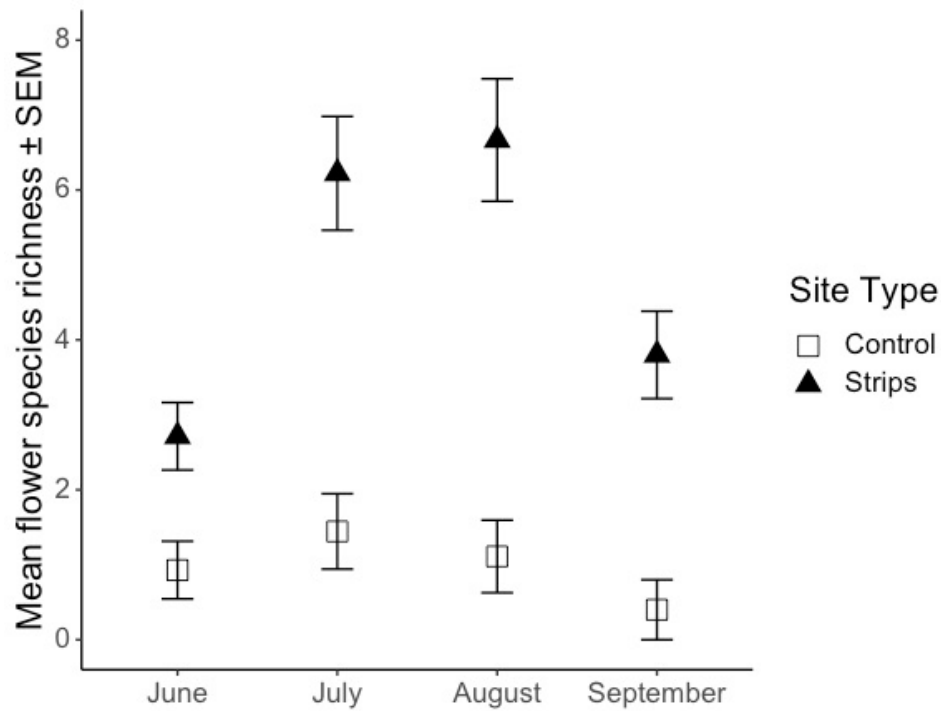


Figure 1.1a. Mean (\pm SEM) number of plant species with flowers for prairie strips sites (triangle points) and control sites (square points) in central Iowa during 2018 and 2019. Blooming flower surveys were conducted once per month during June – August of 2018, and in 2019 we added an additional survey date in May, and another in September.

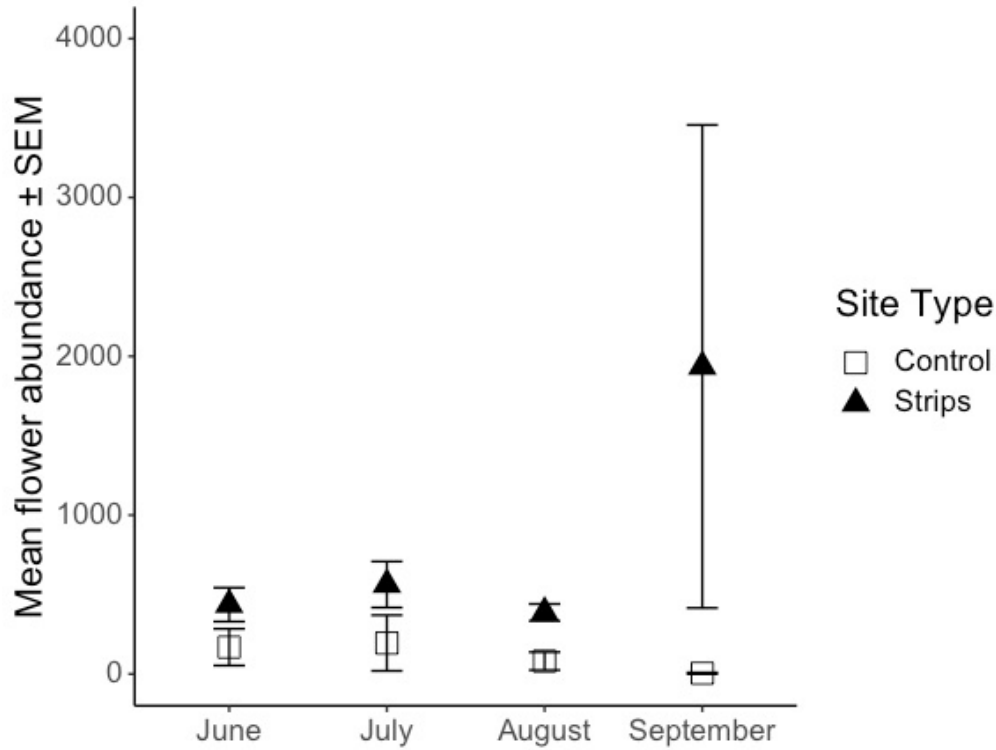


Figure 1.1b. Mean (\pm SEM) number of number of flowers for prairie strips sites (triangle points) and control sites (square points) in central Iowa during 2018 and 2019. Blooming flower surveys were conducted once per month during June – August of 2018, and in 2019 we added an additional survey date in May, and another in September.

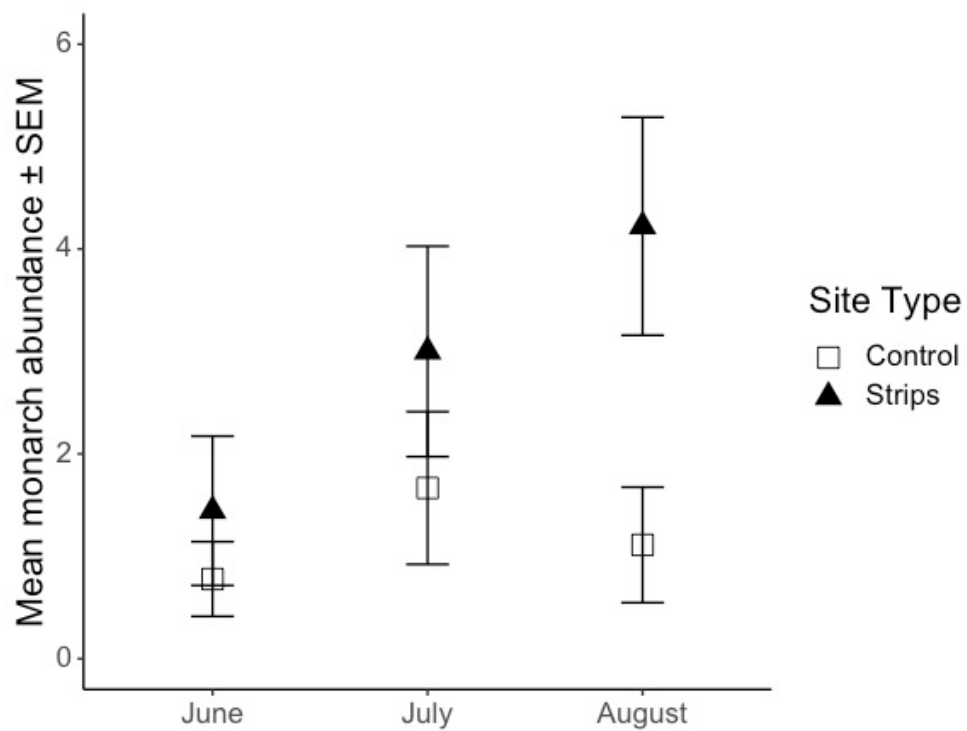


Figure 1.2. Mean (\pm SEM) adult monarch butterflies observed during Pollard walks at prairie strips sites (triangle points) and control sites (square points) in central Iowa during 2018 and 2019. Adult monarchs were surveyed once per month, June through August.

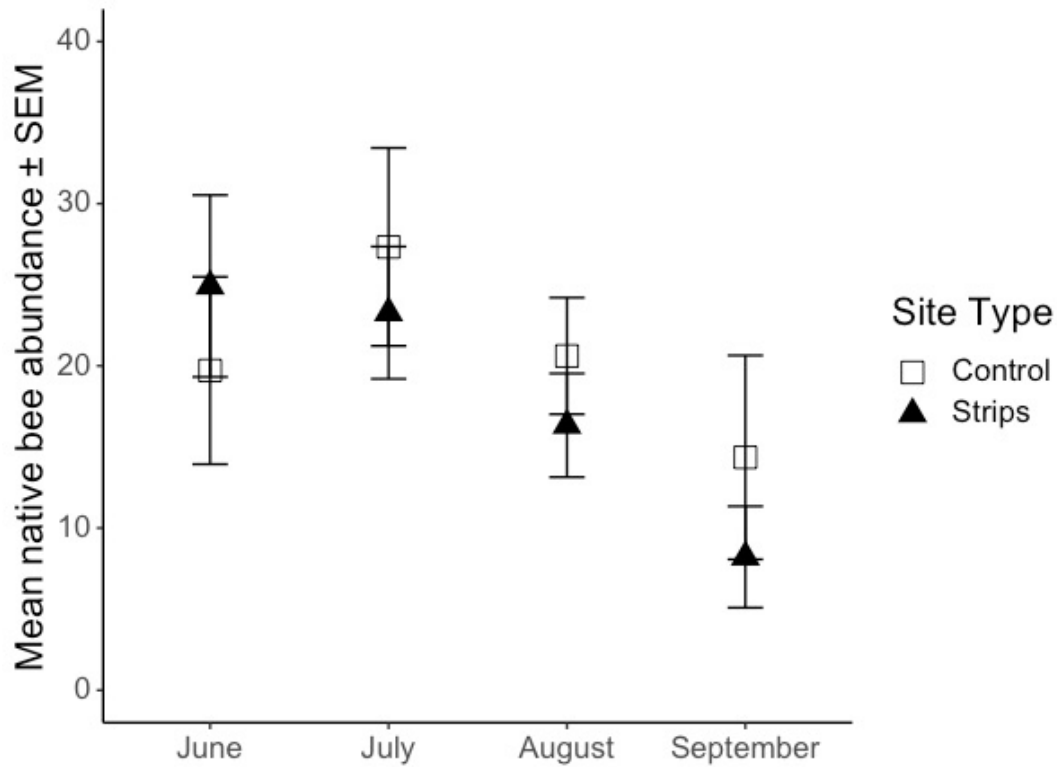


Figure 1.3a. Mean (\pm SEM) abundance of native bees (A) captured in bee bowls at prairie strips sites (triangle points) and control sites (square points) in central Iowa during 2018 and 2019.

Native bee species that varied significantly in activity-density between site types include *A. virescens* (B), *B. pensylvanicus* (B), *H. ligatus* (C), and *M. desponsa* (D). The mean represents the mean number of native bees captured per site per day, as the contents of all bee bowls were combined per site. Bee bowls were deployed twice per month from June until August, and once monthly in September and October. Results were binned by month (October combined with September) for analysis and figures.

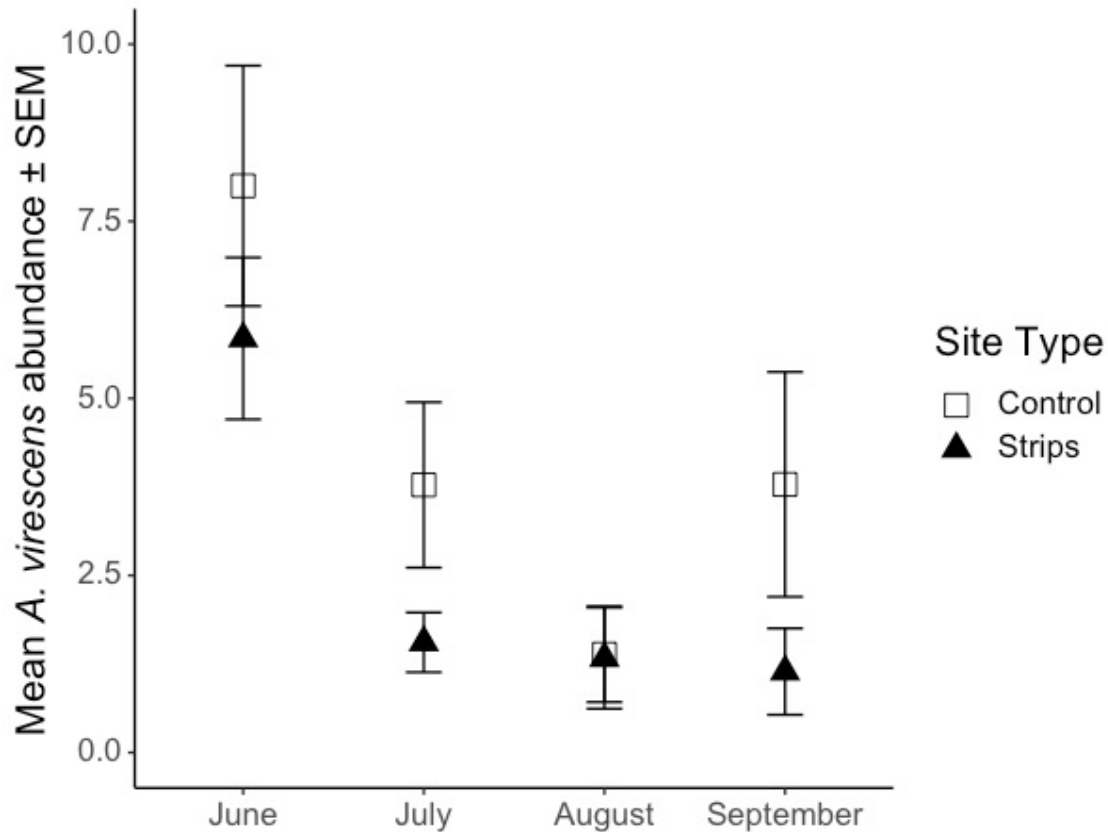


Figure 1.3b. Mean (\pm SEM) abundance of *A. virescens* captured in bee bowls at prairie strips sites (triangle points) and control sites (square points) in central Iowa during 2018 and 2019. The mean represents the mean number of native bees captured per site per day, as the contents of all bee bowls were combined per site. Bee bowls were deployed twice per month from June until August, and once monthly in September and October. Results were binned by month (October combined with September) for analysis and figures.

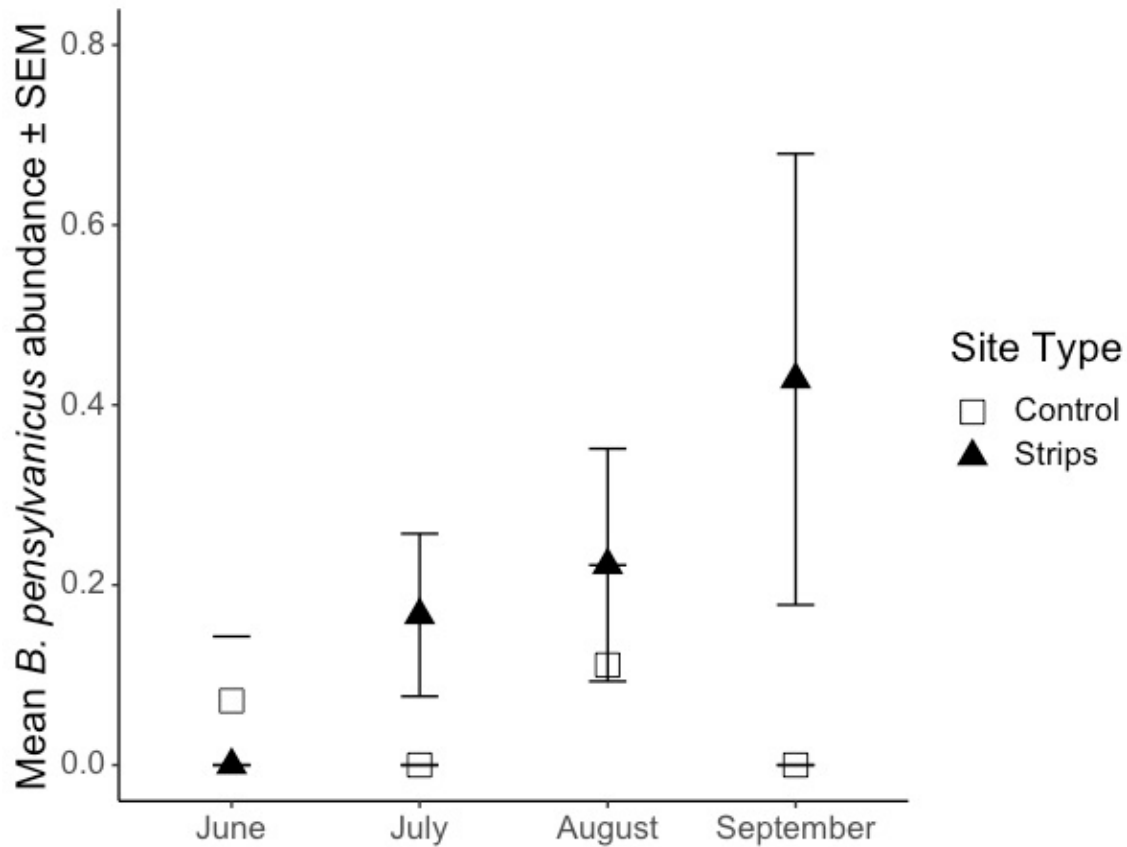


Figure 1.3c. Mean (\pm SEM) abundance of *B. pensylvanicus* captured in bee bowls at prairie strips sites (triangle points) and control sites (square points) in central Iowa during 2018 and 2019. The mean represents the mean number of native bees captured per site per day, as the contents of all bee bowls were combined per site. Bee bowls were deployed twice per month from June until August, and once monthly in September and October. Results were binned by month (October combined with September) for analysis and figures.

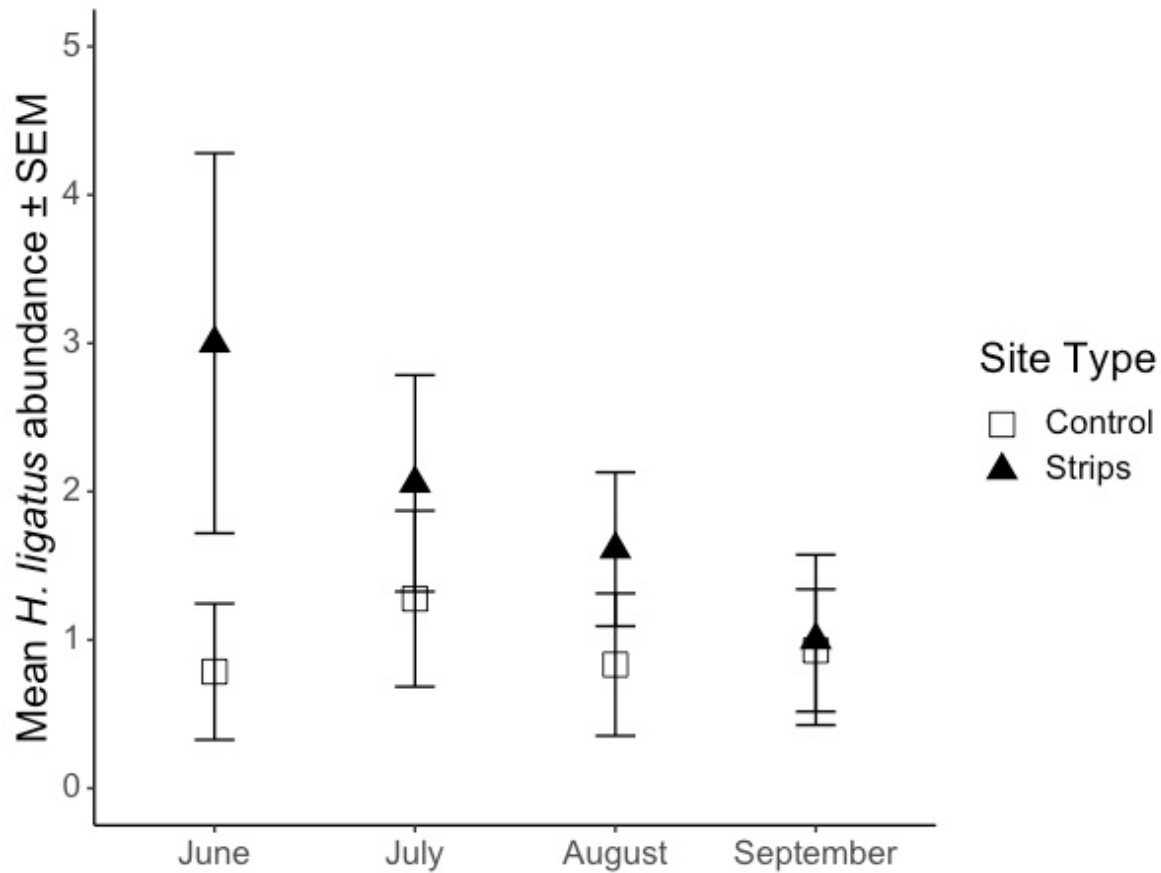


Figure 1.3d. Mean (\pm SEM) abundance of *H. ligatus* captured in bee bowls at prairie strips sites (triangle points) and control sites (square points) in central Iowa during 2018 and 2019. The mean represents the mean number of native bees captured per site per day, as the contents of all bee bowls were combined per site. Bee bowls were deployed twice per month from June until August, and once monthly in September and October. Results were binned by month (October combined with September) for analysis and figures.

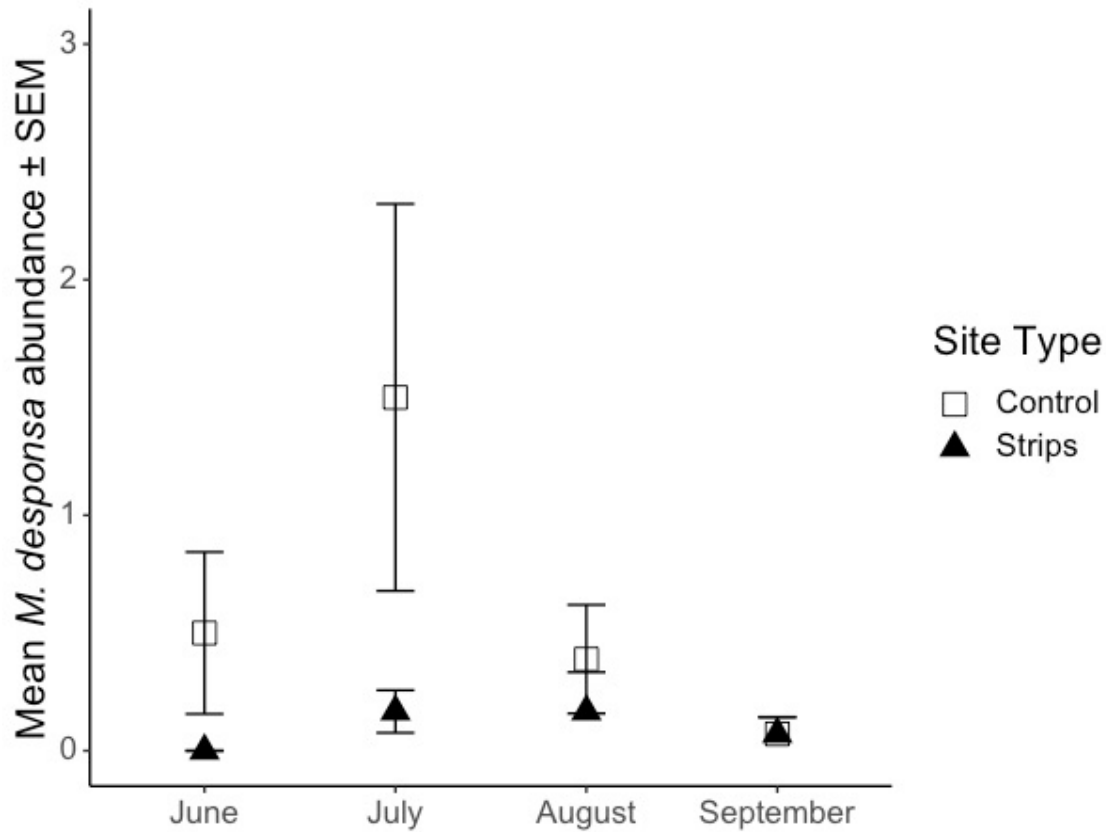


Figure 1.3e. Mean (\pm SEM) abundance of *M. desponsa* captured in bee bowls at prairie strips sites (triangle points) and control sites (square points) in central Iowa during 2018 and 2019. The mean represents the mean number of native bees captured per site per day, as the contents of all bee bowls were combined per site. Bee bowls were deployed twice per month from June until August, and once monthly in September and October. Results were binned by month (October combined with September) for analysis and figures.

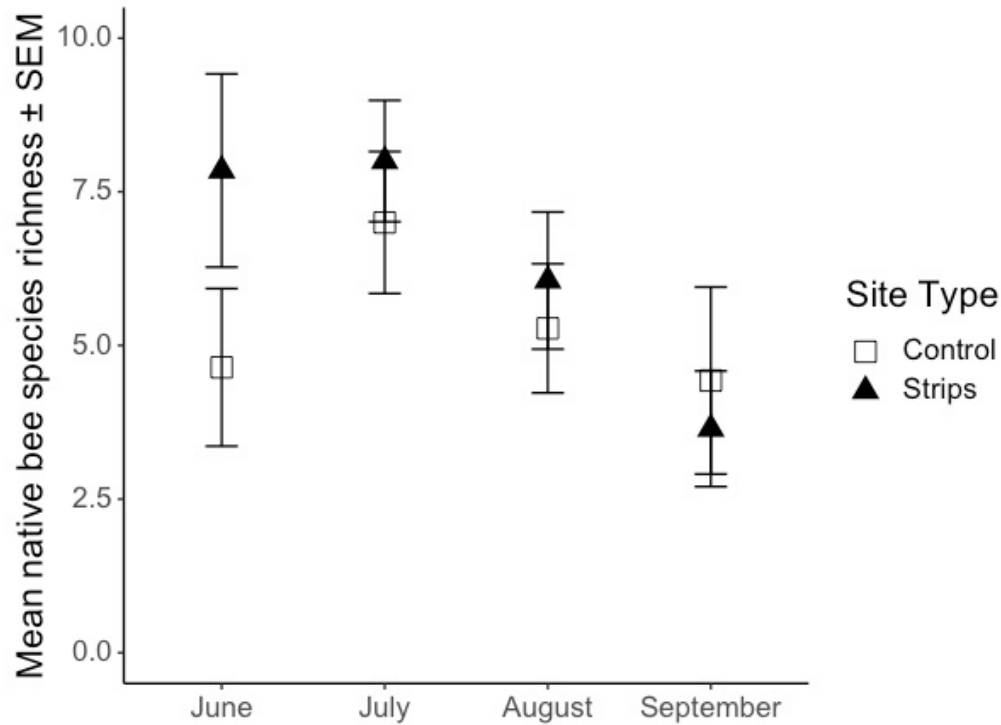


Figure 1.4. Mean (\pm SEM) native bee species richness as estimated by adults captured in bee bowls at prairie strips sites (triangle points) and control sites (square points) in central Iowa during 2018 and 2019. The mean represents the mean number of native bee species captured per site per day, as the contents of all bee bowls were combined per site. Bee bowls were deployed twice per month from June until August, and once monthly in September and October. Results were binned by month for analysis and figures.

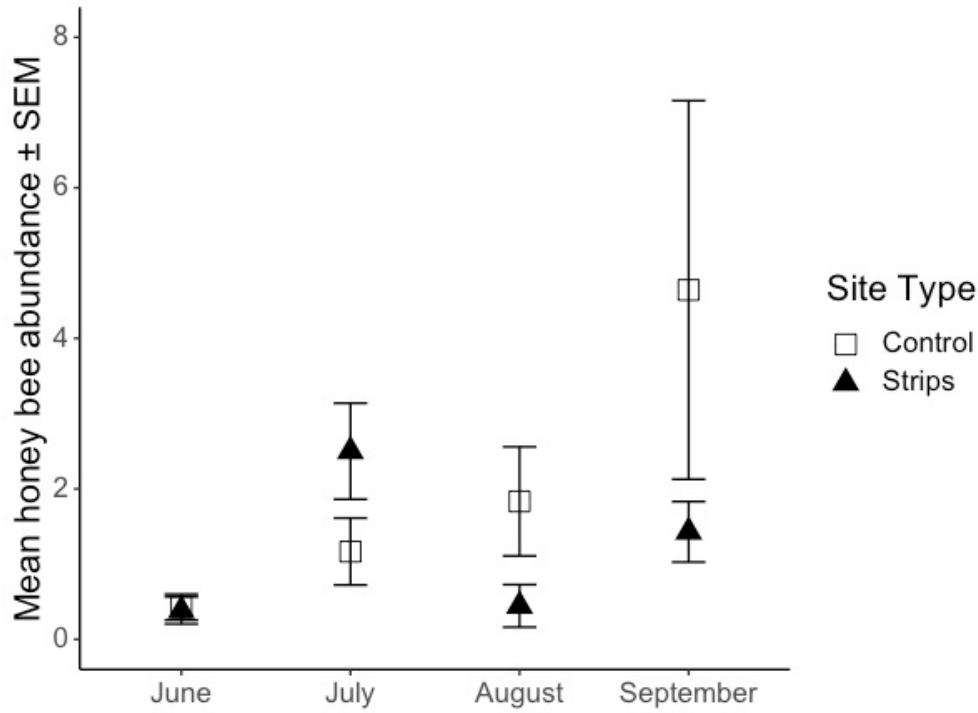


Figure 1.5. Mean (\pm SEM) honey bees captured in bee bowls at prairie strips sites (triangle points) and control sites (square points) in central Iowa during 2018 and 2019. The mean represents the mean number of honey bees captured per site per day, as the contents of all bowls were combined per site. Bee bowls were deployed twice per month from June until August, and once monthly in September and October. Results were binned by month for analysis and figures.

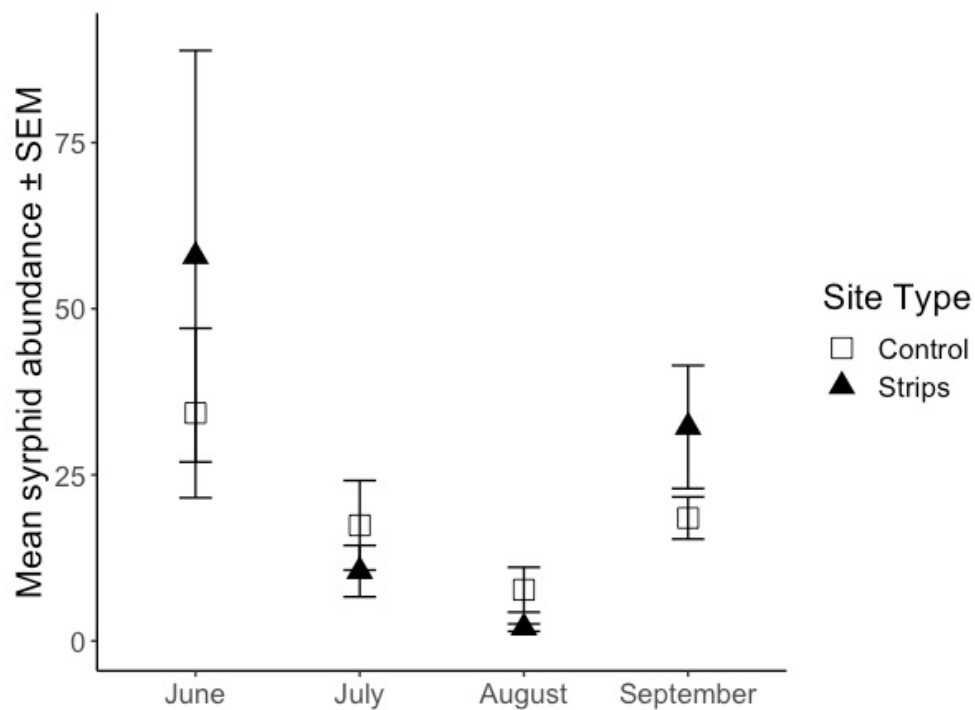


Figure 1.6a. Mean (\pm SEM) abundance of syrphids captured in bee bowls at prairie strips sites (triangle points) and control sites (square points) in central Iowa during 2019. The mean represents the mean number of syrphids captured per site per day, as the contents of all bee bowls were combined per site. Bee bowls were deployed twice per month from June until August, and once monthly in September and October. Results were binned by month for analysis and figures.

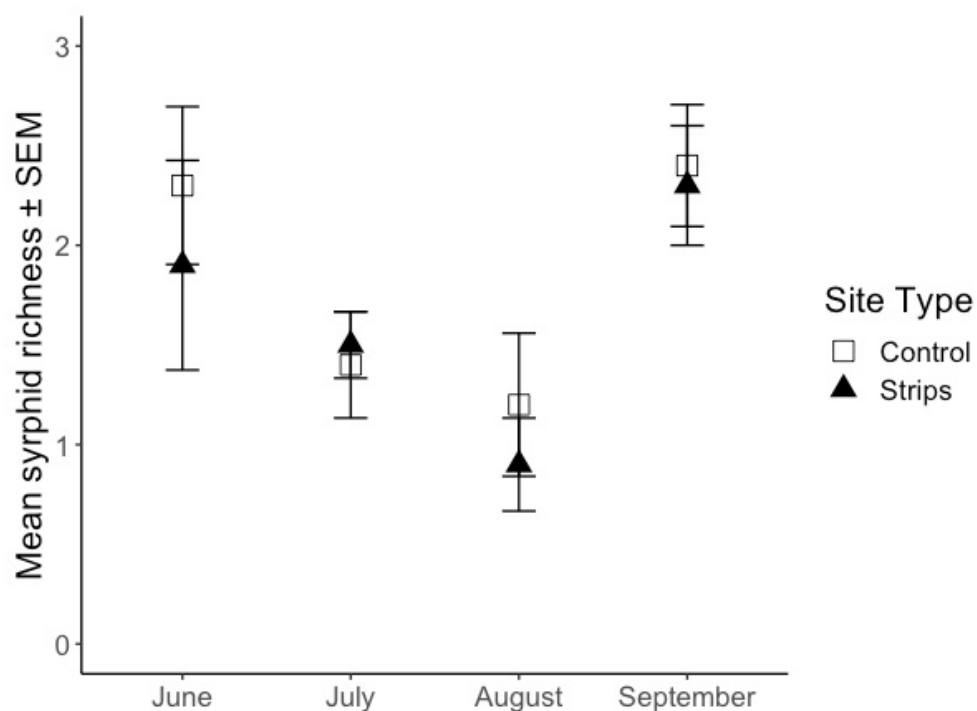


Figure 1.6b. Mean (\pm SEM) syrphid richness of flies captured in bee bowls at prairie strips sites (triangle points) and control sites (square points) in central Iowa during 2019. The mean represents the mean number of syrphid species captured per site per day, as the contents of all bee bowls were combined per site. Bee bowls were deployed twice per month from June until August, and once monthly in September and October. Results were binned by month for analysis and figures.

Table 1.1 Count of native bee individuals collected in bee bowls at each site type listed from most abundant to least abundant within each family.

Family	Genus	Species	Control	Strips	T-test*
Andrenidae	<i>Andrena</i>	<i>crateagi</i>	5	2	NS
	<i>Perdita</i>	<i>halictoides</i>	7	0	NS
	<i>Andrena</i>	<i>krigiana</i>	4	1	NS
	<i>Calliopsis</i>	<i>andreniformis</i>	2	0	NS
	<i>Calliopsis</i>	<i>nebraskensis</i>	0	1	NS
	<i>Pseudoparagus</i>	<i>albitarsis</i>	0	1	NS
	<i>Andrena</i>	<i>canadensis</i>	1	0	NS
	<i>Andrena</i>	<i>cressonii</i>	1	0	NS
	<i>Andrena</i>	<i>geranii</i>	1	0	NS
Apidae	<i>Melissodes</i>	<i>bimaculata</i>	346	198	NS
	<i>Melissodes</i>	<i>trinodis</i>	39	27	NS
	<i>Melissodes</i>	<i>agilis</i>	25	34	NS
	<i>Melissodes</i>	<i>desponsa</i>	42	7	0.04128
	<i>Eucera</i>	<i>hamata</i>	20	4	NS
	<i>Melissodes</i>	<i>communis</i>	8	13	NS
	<i>Melissodes</i>	<i>dentriventis</i>	11	2	NS
	<i>Bombus</i>	<i>pensylvanicus</i>	3	13	0.04938
	<i>Melissodes</i>	<i>niveus</i>	2	3	NS
	<i>Bombus</i>	<i>auricomus</i>	1	3	NS
	<i>Melissodes</i>	<i>druriellus</i>	5	0	NS
	<i>Bombus</i>	<i>griseocollis</i>	2	2	NS
	<i>Nomada</i>	<i>superba</i>	0	3	NS
	<i>Bombus</i>	<i>impatiens</i>	2	1	NS
	<i>Svastra</i>	<i>obliqua</i>	2	1	NS
	<i>Bombus</i>	<i>vagans</i>	0	2	NS
	<i>Bombus</i>	<i>bimaculatus</i>	2	0	NS
	<i>Bombus</i>	<i>fervidus</i>	1	1	NS
	<i>Ceratina</i>	<i>dupla</i>	1	1	NS
	<i>Triepeolus</i>	<i>cressonii</i>	1	1	NS
	<i>Ceratina</i>	<i>mikmaqi</i>	0	1	NS
	<i>Melissodes</i>	<i>coreopsis</i>	0	1	NS
	<i>Nomada</i>	<i>vincta</i>	0	1	NS
	<i>Nomada</i>	<i>spp.</i>	1	0	NS

Table 1.1 Continued

Family	Genus	Species	Control	Strips	T-test*
Colletidae	<i>Melissodes</i>	<i>vernoniae</i>	1	0	NS
	<i>Triepeolus</i>	<i>lunatus</i>	1	0	NS
	<i>Hylaeus</i>	<i>messilae</i>	1	1	NS
Halictidae	<i>Agapostemon</i>	<i>virescens</i>	258	144	0.03202
	<i>Halictus</i>	<i>ligatus</i>	62	119	0.04881
	<i>Lasioglossum</i>	<i>sp 25</i>	70	80	NS
	<i>Lasioglossum</i>	<i>sp 37</i>	49	87	NS
	<i>Lasioglossum</i>	<i>sp 14</i>	42	48	NS
	<i>Halictus</i>	<i>confusus</i>	37	51	NS
	<i>Agapostemon</i>	<i>texanus</i>	49	26	NS
	<i>Lasioglossum</i>	<i>sp 17</i>	34	36	NS
	<i>Augochlorella</i>	<i>aurata</i>	27	41	NS
	<i>Lasioglossum</i>	<i>sp 18</i>	15	43	NS
	<i>Lasioglossum</i>	<i>sp 16</i>	19	20	NS
	<i>Lasioglossum</i>	<i>sp 3</i>	9	14	NS
	<i>Halictus</i>	<i>parallelus</i>	10	13	NS
	<i>Lasioglossum</i>	<i>sp 12</i>	12	11	NS
	<i>Halictus</i>	<i>rubicundis</i>	5	17	NS
	<i>Lasioglossum</i>	<i>sp 9</i>	8	14	NS
	<i>Lasioglossum</i>	<i>sp 11</i>	12	5	NS
	<i>Nomia</i>	<i>universitatis</i>	13	2	NS
	<i>Augochlorella</i>	<i>pura</i>	5	9	NS
	<i>Lasioglossum</i>	<i>sp 31</i>	6	8	NS
	<i>Lasioglossum</i>	<i>sp 5</i>	7	7	NS
	<i>Lasioglossum</i>	<i>sp 6</i>	10	3	NS
	<i>Lasioglossum</i>	<i>sp 38</i>	6	5	NS
	<i>Lasioglossum</i>	<i>sp 28</i>	2	6	NS
	<i>Lasioglossum</i>	<i>evylaeus</i>	8	0	NS
	<i>Lasioglossum</i>	<i>sp 39</i>	2	2	NS
	<i>Sphecodes</i>	<i>spp.</i>	1	2	NS
	<i>Lasioglossum</i>	<i>sp 2</i>	1	2	NS
	<i>Lasioglossum</i>	<i>sp 19</i>	1	2	NS
	<i>Lasioglossum</i>	<i>sp 32</i>	1	2	NS
	<i>Augochloropsis</i>	<i>metallica</i>	2	0	NS
	<i>Duforea</i>	<i>novaeangliae</i>	2	0	NS
	<i>Lasioglossum</i>	<i>sp 24</i>	2	0	NS

Table 1.1 Continued

Family	Genus	Species	Control	Strips	T-test*
	<i>Dieunomia</i>	<i>triangulifera</i>	0	1	NS
	<i>Halictus</i>	<i>tripartitus</i>	0	1	NS
	<i>Lasioglossum</i>	<i>sp 29</i>	0	1	NS
	<i>Lasioglossum</i>	<i>sp 36</i>	0	1	NS
	<i>Lasioglossum</i>	<i>sp 8</i>	1	0	NS
	<i>Lasioglossum</i>	<i>sp 10</i>	1	0	NS
	<i>Lasioglossum</i>	<i>sp 13</i>	1	0	NS
	<i>Lasioglossum</i>	<i>sp 20</i>	1	0	NS
	<i>Lasioglossum</i>	<i>sp 41</i>	1	0	NS
Megachilidae	<i>Megachile</i>	<i>latimanus</i>	8	2	NS
	<i>Megachile</i>	<i>pugnata</i>	0	1	NS
	<i>Megachile</i>	<i>brevis</i>	1	1	NS
	<i>Megachile</i>	<i>relativa</i>	1	0	NS
<i>Total native bees</i>			1340	1152	NS

* values in this column represent the p-value results from each t-test. NS signifies “not significant”

** All *Lasioglossum* are from subgenus *dialictus*.

Table 1.2 Count of Syrphid individuals collected in bee bowls at each site type listed from most abundant to least abundant within each family. The results of a t-test comparing each species' mean abundance between site types are included.

Family	Genus	Species	Control	Strips	T-test
Syrphidae	<i>Toxomerus</i>	<i>marginatus</i>	706	933	NS
	<i>Toxomerus</i>	<i>geminatus</i>	35	33	NS
	<i>Helophilus</i>	<i>latifrons</i>	10	25	NS
	<i>Sphaerophoria</i>	<i>contigua</i>	6	18	NS
	<i>Syrphinae</i>	<i>spp</i>	10	5	NS
	<i>Eristalis</i>	<i>stipator</i>	2	9	NS
	<i>Allograpta</i>	<i>obliqua</i>	5	0	NS
	<i>Eupeodes</i>	<i>volucris</i>	1	2	NS
	<i>Toxomerus</i>	<i>politis</i>	2	1	NS
	<i>Melanostoma</i>	<i>mellinum</i>	2	0	NS
	Total syrphid flies		779	1026	NS

Table S1. Field sites chosen for all pollinator observations in Iowa in 2018 and 2019.

Site	Site Type	County	Year	Field Placement
GUT	Prairie strips	Story	2018, 2019	Within field
SME	Prairie strips	Webster	2018, 2019	Between field and stream edge
SMI	Prairie strips	Wright	2018, 2019	Between field and stream edge
STN	Prairie strips	Tama	2018, 2019	Within field
WHO	Prairie strips	Story	2019	Within field
HAR	Control	Story	2018, 2019	Grass waterway, within field
JER	Control	Wright	2018, 2019	Road right-of-way, field edge
KOE	Control	Webster	2018, 2019	Road right-of-way, field edge
HER	Control	Tama	2018, 2019	Road right-of-way, field edge
DAI	Control	Story	2019	Road right-of-way, field edge

CHAPTER 3: ANALYZING LANDSCAPE LEVEL LAND USE TO ASSESS THE IMPACT OF PRAIRIE STRIPS ON NATIVE POLLINATORS

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Abstract

Habitat loss and fragmentation are increasingly cited as contributing factors in the decline of certain pollinator taxa. In Iowa, the homogenization of agricultural land over the last several decades has led to a majority of land used for corn and soy production. Reintroducing biodiversity through conservation is one possible solution to reverse the current pollinator decline trend. In our three-part study, we first assessed land cover around several Conservation Reserve Program’s CP-43 (prairie strips) sites to complement the analysis of a pollinator field study. We broadened the scope of our land use analysis by assessing the presence of CP-42 (pollinator habitat) within the foraging range of a subset of bees (Hymenoptera: Apoidea) at the same sites. Finally, we summarized CP-42 patch connectivity on the county and state level to better understand how CP-43 fits into a network of native tallgrass prairie patches in Iowa. There was a higher proportion of corn and soybean land cover surrounding control field sites at the 100 m, 300 m and 750 m foraging distances. Between 1000 m and 3000 m from the bee bowls, we found no differences in the number of patches or area of CP-42 surrounding our research sites. These results suggest that differences in abundance and richness observed between prairie strips and control sites were likely not impacted by other CRP practices on the landscape. Finally, our statewide analysis of CP-42 patch connectivity indicated that only 0.015% of patches were

within 500 m of one another. The network of CP-42 in Iowa is fragmented with connectivity at distances greater than most native bees foraging distances suggesting that impacts of CP-42 on native bees may be highly localized. Our results suggest that to reverse pollinator decline through habitat conservation, we will need to augment extant conservation reserve program funded programs to include additional best management practices.

Key Words

Pollinators, native bees, conservation reserve program

Introduction

Insect pollinators including native bees [Hymenoptera: Apoidea] and managed honey bees [Hymenoptera: Apidae: *Apis mellifera*] are declining in North America (Gixti et al. 2009, Cameron et al. 2011, Steinhauer et al. 2014). A primary cause cited for pollinator decline is habitat loss due to the intensification of agricultural systems in the form of land conversion to row-crop systems (Kremen et al. 2007, Koh et al. 2016, Plourde et al. 2013). In 2017, 64.98% of land acres in Iowa were planted in corn and soy alone, a 2.16% increase from 2002 (NASS-USDA 2017, 2002).

Restoration of pollinator forage and habitat is critical to their conservation. One method of habitat restoration is the strategic incorporation of relatively small amounts of native tallgrass prairie into corn and soybean fields. A conservation practice called prairie strips, designated CP-43 by the USDA's Conservation Reserve Program (CRP), is a version of this method. For CP-43, farmers plant strips of native tallgrass vegetation on ten percent or less of their crop field along the contours or streambanks (USDA FSA 2019). Once established, prairie strips intercept field runoff, thereby reducing soil erosion and nutrient loss, all while creating insect pollinator habitat

(Schulte et al. 2017, Gutierrez-Lopez et al. 2014, Kordbacheh et al. 2020). A more established USDA conservation program is Conservation Practice 42: Pollinator Habitat (CP-42), which can be planted in strips or blocks with a minimum size of one half of an acre and are comprised of a high-diversity perennial tallgrass prairie mix (USDA 2008). Two distinct differences between these two conservation practices are rate of adoption by farmers and target goal of the practice. CP-42 is a widely adopted conservation practice in Iowa that was established in 2008 and specifically targets pollinators (Jones Ritten et al. 2017). In contrast, CP-43 was officially listed as a conservation practice by the USDA in 2019 (USDA FSA 2018), making it less adopted as of 2019, and was designed primarily to improve water quality. Both practices follow similar minimum seed mix diversity requirements. CP-42 requires three native flowering plants per bloom period while CP-43 only requires two per bloom period (USDA NRCS 2015). We conducted this study to determine how spatially interconnected these two conservation practices are on the landscape to determine if pollinator community responses observed in prairie strips could be attributed to a larger matrix of diverse, native perennial habitat.

Due to the field-scale implementation of prairie strips, we used individual, commercial farms committed to the conventional production of corn and soybeans (i.e. not certified organic) as our experimental unit. Fields with prairie strips (prairie strips sites) were selected if they had prairie strips that were established for at least two years. Control sites were road rights-of-way or grass waterways directly adjacent to commercial corn or soybean fields that had no prairie strips incorporated into the fields. All study locations had honey bee apiaries of equal size onsite. Variation in pollinator community abundance and diversity was explored with land use land cover (LULC) data from the USDA and aerial imagery interpretation for both crop production and conservation practices (CP-42 and 43) for these study sites.

Methods

In this study, we used ArcGIS Pro software (Environmental Systems Research Institute, Redlands, CA) to do our spatial analysis for the smaller two spatial extents (0 – 3000 m). For our county-scale analysis of CP-42 connectivity, we used FRAGSTATS, a software designed to perform patch analyses in ecological studies (McGarigal et al. 2002). We used the most recent aerial imagery available at the time of this study (2017 Natural Color Imagery) from the Iowa Geographic Map Server as base maps to digitize land use (GISU 2019).

In a concurrent study using the same research sites, we measured native bee and syrphid fly richness and activity-density as well as honey bee activity-density. In this study, we used geospatial data in a GIS to analyze the potential impact of landscape-level LULC on the observed insect pollinator activity-density and richness.

We analyzed landscape level LULC data on three different scales, using three different methodologies to answer our research question. The first scale was 0 to 750 m from the bee bowl locations at our research sites. This was the smallest of three scales and represents the foraging distance of many small and medium-sized native bees that we collected in our bee bowls (Wright et al. 2015). At this spatial extent, the cover of semi-natural habitat has been positively correlated with native, solitary bee richness and abundance (Steffan-Dewenter et al. 2002).

The second spatial extent was constrained to a 1000 – 3000 m radius from the bee bowl locations to capture foraging distances of larger native bees (i.e. *Bombus*) (Dramstead 1996) and honey bees (Beekman & Ratniecks 2000) (Fig 1). At this scale, we used geospatial data provided by the United States Department of Agriculture (USDA) to analyze the presence of CP-42 within the distances described above. Our goal was to determine if pollinator activity-density and richness differences between site types could be explained by the presence of pollinator habitat

on the landscape scale. We used CP-42 data for this study because it has been established for 13 years in the state of Iowa, and while many conservation practices contribute to pollinator habitat (e.g., filter strips, hedgerow plantings, and wetland restoration (USDA 2015), CP-42 is the only conservation practice designed explicitly for pollinators making it a logical focal point for this study (USDA FSA 2018).

Our final geospatial scale assessed CP-42 area and connectivity at the county and state level. At this scale, we hypothesized that prairie strips fill a conservation connectivity gap for pollinators in the wider Iowa landscape. We assessed CP-42 patch connectivity in the same four Iowa counties as our pollinator study site locations.

Determining foraging distances

We chose three buffer distances (100 m, 300 m, and 750 m) for this study based on the approximate foraging distances of the smallest and largest native bees that were caught in bee bowls. The body length of the smallest bee, a morphospecies of *Lasioglossum dialictus*, was on average, 3.7 mm, and the largest species were *Bombus pensylvanicus* and *Bombus auricomus*, both with average queen body lengths of 25 mm (Mitchell 1962). Wright et al. (2015) found that small bees (<1.5mm intertegular span) foraged below 350 m from their nests approximately 90% of the time. On the larger end of the spectrum, bumble bees forage at least a few hundred meters (Dramstad 1996, Osborne et al. 1999), with another study suggesting a foraging distance further than 1500 m (Kreyer et al. 2004). Foraging distance is correlated non-linearly with the intertegular span of native bees (Greenleaf et al. 2007) which allowed us to estimate foraging distances of lesser studied native bees. Using foraging distances of the smallest and largest bees in our study, we were able to capture the most species' estimated foraging distances.

Land use classifications

To classify land cover, we created a point feature class using the location of the bee bowls at both control and prairie strips sites. We then used the buffer routine in ArcGIS Pro to create 750 m buffers around each bee bowl location. Resulting buffers served as focal areas for manual image interpretation and vector digitization of the LULC types (*sensu* Hines and Hendrix 2005) described below. We consulted contemporary research in pollinator dynamics (Samuelson and Leadbeater 2018, Hines and Hendrix 2005) to inform our choices of LULC types most likely to be relevant to pollinators. We classified all LULC polygons into nine classes (Table 2) that were ground-verified during pollinator surveys in 2019. The LULC designations consisted of agricultural row crops (i.e. corn and soy), domestic (i.e. houses, barns, driveways), forest, grass waterway, open (i.e. pasture), prairie strip, road, road right-of-way, and water.

For each smaller foraging distance, we created buffers (radii 100 m and 300 m) around the bee bowl point locations, and then clipped our classified LULC data using these buffers to maintain the original land use designations.

Land cover analysis

For each foraging distance buffer area (100 m, 300 m, & 750 m), we calculated the mean percent cover of each LULC type and the standard error (%) at each site type (prairie strip and control). We did this by calculating the area of each LULC type in a buffer and dividing it by the total buffer area. We checked for normality using both the Wilks-Shapiro test and visual inspections of plotted data. We then conducted Welch's t-tests (assuming unequal variance) to

determine if there was a significant difference in the percent cover of any land use type surrounding control versus prairie strip research locations (St. Clair et al. 2020).

CP-42 analysis methods

We used 2020 USDA vector data containing all CP-42 patches in Iowa to do analyses at the next largest spatial extent. We first created 1000 m and 3000 m buffers around the bee bowl locations at both prairie strip and control sites in ArcGIS Pro. These buffers represent the area where honey bees engage in approximately 90% of their pollen foraging. (Danner et al. 2016). Bumble bees and other larger native bee species forage within this range as well (Kreyer et al. 2004). To find the number of CP-42 patches and the sum of CP-42 acres within those radii, we clipped the CP-42 polygon feature class using 1000 m and 3000 m buffers. Any CP-42 on the border of these radii were split to include only the area within the buffer distance. We then calculated areas and number of patches of CP-42 land available to pollinators within each study distance.

Patch metrics of CP-42

We used FRAGSTATS (McGarigal et al. 2002) to perform class level analyses of CP-42 patch connectivity on the county scale. Our initial site selection was based on county, thus we felt it appropriate to use the same delineation for our most coarse spatial grain size. Here again, we used 2020 USDA CP-42 vector data for our analysis. We converted the data to raster in ArcGIS Pro to be compatible with the FRAGSTATS software. The county-level CP-42 vector data was converted to raster format using a five-meter output cell size to preserve the character of the input polygon boundaries. Due to file size restrictions, however, the statewide analysis

was performed using a 20-meter output cell size. We calculated mean Euclidean nearest neighbor (EMM), connectivity at 500 m (CONNECT), patch area, and patch number using a “No sampling” strategy in FRAGSTATS (McGarigal et al. 2002). The parameter CONNECT requires an input distance to calculate the percentage of patches that are within the set distance from one another. We chose a 500 m distance to represent many of the native bee species from our study based on approximate foraging distances (Gathmann and Tschardt 2002, Greenleaf et al. 2007).

Results

Land cover at foraging distances

At both the 100 m ($t = 2.5772$, $df = 6.2728$, $p = 0.04029$) and 300m ($t = 3.1121$, $df = 6.7877$, $p = 0.01772$) foraging distance, there was a higher proportions of agriculture (corn and soy) surrounding control sites (Table 1a and 1b). Within 100 m of prairie strips sites there was a marginally higher proportion ($t = -2.273$, $df = 7.8342$, $p = 0.05332$) of grass waterway cover within the same distance of control sites. At prairie strips sites, the prairie strips themselves represented 14.49% (Table 1a) of the total land cover at that foraging distance.

Within 750 m of the sites, there was a marginally higher proportion of agriculture at control sites ($t = 2.2985$, $df = 7.0452$, $p = 0.05488$, Table 1c) than prairie strips sites. There was also a marginally higher proportion ($t = -2.4693$, $df = 5.2024$, $p = 0.05468$) of water cover at prairie strips sites. Prairie strips comprised 2.46% on average of the 750 m radius at prairie strips sites (Table 1c).

CP-42 around prairie strips sites

At the 1000 m level around the field sites, there was only one prairie strips site and one control site where a single CP-42 patch was present. There were no other CP-42 patches within 1000m of our field sites. There were more CP-42 patches present within 3000 m of the field sites, however neither the number of patches nor sum of CP-42 area differed between site types (Table 2).

County scale CP-42 analysis

Of the four counties in Iowa where we had field sites, Webster County had the fewest CP-42 patches per county than the statewide average (50 vs 191 ± 15.33) and the lowest average patch size (6.53 ± 1.23 acres, Table 3). Tama County had the second lowest average patch size but boasted the highest connectivity of CP-42 patches within 500 m, at 2.00% (Table 3). Across the entire state, connectivity within 500 m was a low 0.01%, yet the Euclidean nearest neighbor distance was 588 m, shorter than three of the four counties studied (Table 3).

Discussion

Distinguishing the effects of pollinator conservation on the landscape scale is difficult due to potential confounding variables, such as other pockets of floral resources within the foraging range of most native bees. Landscape effects can be partially determined by LULC classification and analysis. However, there are still potentially impactful characteristics at a fine-grained, less detectable scale such as home gardens. There is also a temporal effect on the quality of pollinator habitat in certain land use categories. For example, row-crop agriculture could be classified as low quality pollinator habitat (Hines and Hendrix 2005), but for several weeks, there

could be a flower explosion, such as the soybean bloom. Studies have found that bees will travel further for large clusters of flowers (Akter et al. 2017) or during forage scarcity (Steffan-Dewenter and Kuhn 2003) so the surrounding land use can significantly impact the dispersion of bees. Our results of the full land use categorization study brought into question potential underlying landscape effects on our ecological native bee survey findings at each buffer (100 m, 300 m, and 750 m). There were significant differences between several categories of LULC (i.e. agriculture and grass waterway) between prairie strips and control sites at both the 100 m and 300 m buffer distances. Small solitary bees are most likely to be impacted by these differences due to their short foraging distances. At the 750 m distance, there was a marginally higher proportion of agriculture around control sites, and a higher proportion of water around prairie strips sites. Water is not likely to be a contributing factor in any pollinator community results that we observed. Our native bee richness data should be further analyzed to determine if the smallest observed native bee species show significant differences in activity-density between site types.

Prairie strips are not the only conservation practice being implementing in Iowa crop fields. By analyzing the presence of CP-42 on the landscape we were able to further understand the potential impact of USDA conservation programs on pollinators. At the spatial extent between 1000 and 3000 m, elevated relative CP-42 connectivity could potentially benefit bees with longer foraging distances like bumble bees and honey bees. While we did find CP-42 within 3000 m of our field sites, there was not a significant difference in the number of CP-42 patches or amount of area between site types. Therefore, pollinators at both site types would be impacted on a similar level by surrounding CP-42 areas. By performing these analyses, we began to

observe potential ways in which a network of targeted CP-42 conservation practices could be placed across an agricultural landscape, which led us to our coarsest scale of investigation.

Such conservation habitat networks are not created equally across Iowa. We saw notable differences between counties in number, area, and connectivity of CP-42 patches, however, we must consider the anthropocentric nature of our spatial delineations. Conservation planning is determined by those delineations, both county and state, but pollinators are not confined by them. Statewide, only 0.01% of CP-42 patches were within 500 m of each another. Even with Iowa currently at full or near full CP-42 enrollment, the connectivity of patches still did not reach the threshold for most native bee foraging distances, making them completely reliant on other habitat to forage in the matrix. In addition, Jauker et al (2009) found that native bee abundance was lower when patches of natural area were isolated. There are obvious limitations to farm-bill funded pollinator habitat which further exemplifies the need for voluntary conservation implementation.

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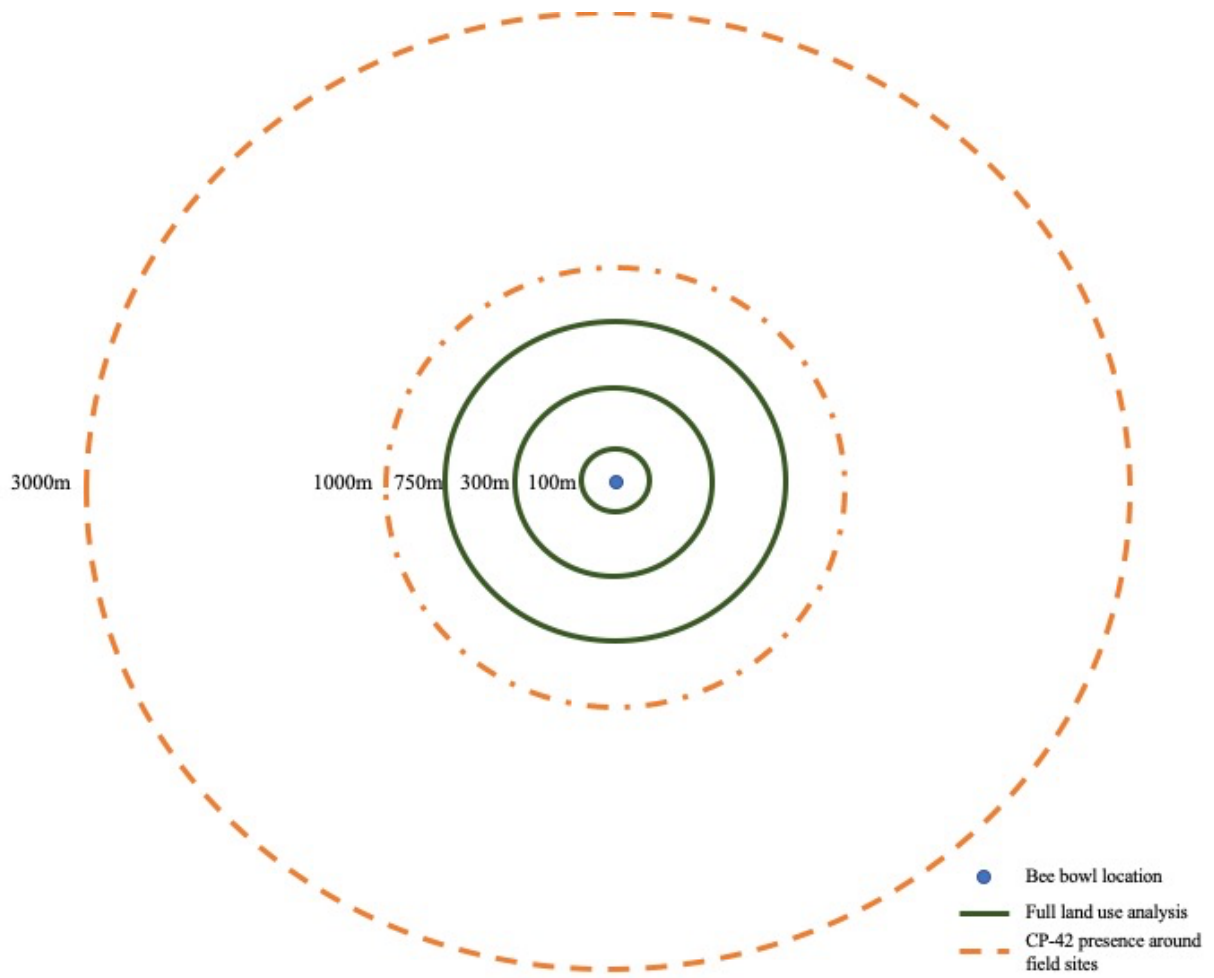
Figures and Tables

Figure 2.1. Diagram depicting scales of two (0 - 750 m, and 1000 - 3000 m) distances from field sites) of the methodologies described in this chapter.

Table 2.1a. Land use land cover percentages surrounding prairie strips and control sites at 100 m. Significance of t-test denoted with an asterisk.

	Prairie strips sites (%)	Control sites (%)	p-value
Agricultural (corn & soy)	55.00 ± 8.15	79.05 ± 4.55	0.040*
Domestic (homesites)	0	4.77 ± 4.77	0.374
Forest	0.06 ± 0.06	0	0.374
Grass waterway	25.42 ± 5.78	8.07 ± 4.99	0.053*
Open (pasture)	0	0	NA
Prairie Strip	14.49 ± 4.17	0	0.025*
Road	1.31 ± 0.83	3.14 ± 1.34	0.287
Road Right-of-Way	1.21 ± 1.21	3.81 ± 1.58	0.23
Water	2.49 ± 2.02	1.17 ± 0.99	0.578

Table 2.1b. Land use land cover percentages surrounding prairie strips and control sites at 300 m. Significance of t-test denoted with an asterisk.

	Prairie strips sites (%)	Control sites (%)	p-value
Agricultural (corn & soy)	69.22 ± 5.10	88.03 ± 3.25	0.018*
Domestic (homesites)	1.42 ± 0.85	2.62 ± 1.03	0.392
Forest	2.08 ± 2.08	0.11 ± 0.113	0.398
Grass waterway	14.81 ± 3.87	4.39 ± 2.86	0.065
Open (pasture)	1.70 ± 1.06	0	0.183
Prairie Strip	5.57 ± 2.40	0	0.081
Road	1.24 ± 0.59	1.86 ± 0.49	0.447
Road Right-of-Way	1.63 ± 0.70	2.56 ± 0.68	0.371
Water	1.07 ± 0.69	0.43 ± 0.32	0.439

Table 2.1c. Land use land cover percentages surrounding prairie strips and control sites at 750

m. Significance of t-test denoted with an asterisk.

	Prairie strips sites (%)	Control sites (%)	p-value
Agricultural (corn & soy)	73.49 ± 4.44	85.84 ± 3.02	0.055*
Domestic (homesites)	5.40 ± 2.52	5.35 ± 2.21	0.988
Forest	4.12 ± 2.52	1.65 ± 1.65	0.439
Grass waterway	6.64 ± 1.78	3.43 ± 2.20	0.291
Open (pasture)	2.74 ± 1.60	0.41 ± 0.41	0.222
Prairie Strip	2.46 ± 1.09	0	0.087
Road	1.82 ± 0.49	1.12 ± 0.12	0.265
Road Right-of-Way	2.29 ± 0.83	1.95 ± 0.57	0.740
Water	1.03 ± 0.32	0.20 ± 0.12	0.055*

Table 2.2. Pollinator Habitat conservation practice (CP-42) patch area and count at 1000 m and 3000 m distances from bee bowls located in prairie strips.

	Prairie Strips	Control	p-value
1000m Patch Area (ac)	2.82 ± 2.52	1.83 ± 1.63	0.78
3000m Patch Area (ac)	28.82 ± 15.10	54.29 ± 30.13	0.52
1000 Patch count	0.2 ± 0.18	0.2 ± 0.18	1.00
3000 Patch count	4.6 ± 2.73	4.8 ± 2.86	0.96

Table 2.3. Pollinator Habitat conservation practice (CP-42) composition and configuration in Iowa counties

Locality	Number of Patches per County	Mean Patch Area (acres)	Patch Connectivity at 500m (CONNECT)	Mean Euclidean Nearest Neighbor Distance (m)
All IA County Average	191.25 ± 15.33	11.59 ± 0.53	0.015% ¹	587.78 ¹
Story	186	12.16 ± 1.38	0.93%	605.08
Tama	139	11.08 ± 1.41	2.00%	627.78
Webster	360	12.79 ± 1.03	0.56%	441.46
Wright	50	6.53 ± 1.23	0.68%	1923.65

¹Based on statewide calculations, not county averages. Raster cell size used was 20m, due to file size limitations in the FRAGSTATS program. All county level calculations in FRAGSTATS used raster cell size of 5m.

CHAPTER 4: GENERAL CONCLUSION

The broad objective of this thesis was to determine the impact of prairie strips on the abundance and richness of certain pollinator taxa in Iowa corn and soybean fields. I achieved this goal by completing field studies to quantify monarch butterfly, native bee, honey bee, and syrphid presence at control and prairie strips sites. I also surveyed milkweed and flowering plants to detect differences in available critical pollinator resources. Finally, I broadened the scope of the field study by completing a spatial analysis of land use in the areas around our field sites to assess any potential confounding variables. While our primary research was focused on prairie strips, there are many other conservation practices on the Iowa landscape. To address a subset of these alternative conservation practices, I extended the spatial analysis to include pollinator habitat conservation practice (CP-42) patch connectivity on the county and state level.

In Chapter 2, we determined that certain pollinator responses were greater at prairie strips sites (i.e. monarch butterfly abundance) across the entire field season, while some were greater in certain months (native bee activity-density and richness, syrphid activity-density). These observed temporal differences provide valuable insight into the key phenology of perennial vegetation when it may be most critical to pollinators within the Iowa agricultural matrix. We can use our flowering plant data to pinpoint which species were most prevalent during times when pollinators were most active in prairie strips. Combined with recent pollen analyses of honey bee apiaries at prairie strips (Zhang et al. 2020), these data could optimize tallgrass prairie seed-mixes used to establish conservation practices.

Also in Chapter 2, we observed more adult monarchs at prairie strips sites despite finding more milkweed at control sites. Monarch conservationists could apply this finding to habitat restoration efforts. Common milkweed thrives in highly disturbed areas, like our control sites

(i.e. road rights-of-way), and is used readily by monarchs to deposit eggs. Adult monarchs use the high floral abundance and diversity of prairie strips to gather nectar to fuel fall migration. Moving forward, monarch conservation efforts should prioritize both habitat types.

Another notable Chapter 2 finding was that syrphids did not respond consistently to prairie strips in terms of activity density and richness. Syrphids seem ubiquitous in Iowa, and our research results truly capture the remarkable number of syrphids that occupy intense agriculture landscapes. Syrphids, specifically *T. marginatus*, likely did not respond differently to prairie strips during most months due having a more generalist feeding strategy. Future studies should be conducted to determine if there are alternative habitats that increase syrphid activity density.

The geospatial analyses described in Chapter 3, specifically using FRAGSTATS, could be used to determine pollinator habitat fragmentation within a variety of conservation practices. With a comprehensive view of the conservation matrix, habitat restoration could be targeted towards areas with low pollinator habitat patch connectivity.

Overall, the chapters in this thesis report methodology and implications of pollinator response to prairie strips as well as landscape level connectivity of pollinator habitat in Iowa. As prairie strips become more widely adopted, more studies of pollinator response should be done in Iowa and other Midwestern states. The corn and soybean landscape extends beyond Iowa, and pollinator communities differ by geographical location, so their responses should be captured. This thesis provides a baseline knowledge of how certain pollinator taxa respond to prairie strips and can be used by conservationists to encourage further adoption of this conservation practice.

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