

**THE USE OF FLORAL ENHANCEMENTS TO
SUPPORT BENEFICIAL INSECTS IN PRAIRIE
AGROECOSYSTEMS**

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By

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ABSTRACT

THE USE OF FLORAL ENHANCEMENTS TO SUPPORT BENEFICIAL INSECTS ON MANITOBA FARM LAND

By Michael Killewald

Insects worldwide are in decline, with strong evidence that climate change and agricultural practices are to blame. Pesticides, tillage, and land conversion all negatively affect insect community structure. Recent policy changes have not proved effective enough to conserve insects. Understanding long-term techniques to manage and conserve insects in agricultural settings is necessary to maintain the ecosystem services they provide. Floral strips have proven effective in perennial cropping systems, but it is unclear if these practices will prove effective in annual systems. I installed fifteen floral strips adjacent to annual crop fields, and five hay fields planted with a pollinator enhanced seed mix in Manitoba, Canada. I measured these sites for multiple years to determine beneficial insect responses to the enhancement, farm management techniques, and the surrounding landscape. Insect functional traits are investigated and discussed.

I found that floral strips generally increased capture rates of ground beetles (Coleoptera: Carabidae), and bees (Hymenoptera: Apoidea), but not syrphids (Diptera: Syrphidae). Bees spilled over from the strips into the adjacent annual crop fields, but I did not observe this trend for ground beetles or syrphids. Ground beetles responded to the landscape at small (500 m) scales and were positively associated with habitat connectivity. Syrphids were positively associated with landscapes with continuous

patches of landcover at small (500 m) scales. Bees responded at larger (3 km) scales and were positively associated with forb cover and landscape diversity. Floral strips supported the reproduction of bees. I found more stem-nests with a higher dietary diversity at strip sites. Additionally, 23.5% of offspring provisions may have come from the strips. Bee communities differed based on vegetation management. Enhanced hay fields supported fewer bees than grazed or natural sites, and captured primarily honey bees. Grazed fields supported primarily ground-nesting, solitary bees, while natural sites supported more hive-nesting, polylectic bees. Low capture rates of bees in hay fields could be due to their lack of floral diversity, as > 99 % of ground cover was alfalfa. I have shown that floral enhancements can be an effective tool for managing insects in annual cropping systems. Recommendations for their use in Manitoba are discussed.

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DEDICATION

To my parents, Diane and George Killewald, who constantly believed in me; To my younger self, who kept pushing for every accomplishment; To anyone who believes they can survive past chapter 1.

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AUTHOR CONTRIBUTIONS

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MFK designed experiments with support from JG, AC, and YL

MFK performed all field experiments and processed samples with help from JG, AC, and
lab technicians

MFK identified all pollen samples

MFK identified all beetles, most of the Syrphidae, and roughly half of the bees

(particularly *Bombus*, *Megachile*, *Halictus*, *Ceratina*, and species of other groups
where identifications were confident)

JG identified roughly half of the bees (particularly *Lasioglossum* and Andrenidae)

MFK completed drafts of written materials and incorporated edits from JG, AC, YL, and
RG

MFK performed statistical analysis with help from JG

MFK submitted manuscripts for publication with guidance from JG, AC, YL, and RG

JG, AC, YL, and RG provided funding to establish strips used in MFK chapters 3–5

CHAPTER 1. GENERAL INTRODUCTION

1.1 INTRODUCTION

Insect declines have been detected across the globe (Wagner 2020) and rates of declines increase with agricultural presence (Mancini et al. 2023). Given that insects provide many ecosystem services such as pollination (Kremen et al., 2002) and biological control (Bianchi et al., 2006), these declines put the future of agricultural production at risk (van der Sluijs 2020). Efforts to manage beneficial insects in agricultural areas have proven successful in other studies (Blaauw & Isaacs, 2014, 2015; Kordbacheh et al., 2020; N. M. Williams et al., 2024) but overall effects vary between landscapes (Albrecht et al., 2021; Zamorano et al., 2020). Most studies in North America that use habitat enhancements to support insects are adjacent to perennial crops (Zamorano et al. 2020), which provides more resource stability between years than rotational systems. Given the differences in responses between landscapes and the lack of research on the use of floral strips in rotational systems (but see Vickruck et al., 2021), more research is necessary to effectively use this management tool.

The goal of my study was to determine ways to use habitat enhancements and crop management techniques to best conserve insect populations within rotationally managed agricultural systems. I was able to select multiple sites to both establish enhancements and monitor current farm management techniques. I compared these sites

to semi-natural areas and the surrounding landscape to gauge relative insect conservation levels. I was ultimately able to identify key variables that influence insect populations and that aid in their conservation and management in rotational farming systems.

1.2 THESIS ORGANIZATION

The following chapters are written in manuscript format and includes seven chapters, including this introduction as chapter 1. Chapter 2 is a literature review of beneficial insects and common Integrated Pest and Pollinator Management (IPPM) techniques. Chapters 3–6 are data chapters and all of these have either been published (Chapter 3, 4, and 5) or accepted for publication, pending revisions, (Chapter 6).

Chapter 3 analyses the numerical and functional responses of ground beetles, bees, and hover flies to floral strips during the first year of establishment (2019). This chapter is published in the *Journal of Agricultural and Forest Entomology* (Killewald et al. 2024) and includes discussion on abundance, diversity, and functional differences between insect communities at strip, control, and semi-natural sites in Manitoba, Canada. I also examined landscape and farm management effects on insect communities and differences are discussed. In 2019, the strip and control fields were planted with the same crops, allowing for direct comparisons between them. In 2020 onwards, farmers planted different crops in each field. Thus, I separated these data into different chapters to ask different questions.

Chapter 4 analyzes the responses of beneficial insects to floral strips in years 1–3 after establishment. This chapter is published in the *Journal of Agricultural and Forest Entomology* (Killewald et al. 2025) This chapter analyzes the differences in abundance

and diversity of insect communities between strip, control, and semi-natural sites. I also investigated landscape effects and determined how community responses changed over time (years 1–3).

Chapter 5 analyzes the reproductive and dietary responses of leafcutter bees to floral strips adjacent to rotationally managed crop fields. I analyzed how pollen diversity, treatment, and landscape structure influenced nesting rates of leafcutter bees at strip, control, and semi-natural sites in Manitoba, Canada. This manuscript is published with the *Journal of Agriculture, Ecosystems, and the Environment* (Killewald and Gibbs 2025).

The final data chapter of my thesis, Chapter 6, investigates the community differences of bees between grazed, hayed, and semi-natural sites in Manitoba, Canada. I analysed community structure and functional responses of bees to both treatment and landscape context. This manuscript is currently accepted, pending revisions, with *The Canadian Entomologist*.

Finally, chapter 7 synthesizes the preceding data chapters and draws general conclusions from my results.

CHAPTER 2. LITERATURE REVIEW

2.1 INTRODUCTION

This thesis explores the use of Integrated Pest and Pollinator Management (IPPM) techniques in agricultural production areas in the prairie region of Manitoba, Canada. Techniques included implementing floral strips adjacent to rotationally managed crop fields and the use of grazing and haying practices to support beneficial insects. This literature review serves as an introduction to key insect ecosystem services, integrated pest and pollinator management, and the use of habitat enhancements to support beneficial insects in agricultural production.

2.2 BENEFICIAL INSECTS

2.2.1 Introduction to beneficial insects

Beneficial insects are a diverse group that provide a net benefit to humans in some form (Alford 2019). These benefits, and others, have been collectively called “ecosystem services” (Daily et al. 2003). Pollination and biological control are notable insect ecosystem services as they have a direct benefit to pest management (Obrycki et al. 2009), and food (Klein et al. 2007) and seed production (Ollerton et al., 2011). Beneficial insects are also important decomposers (Iqbal et al. 2020), seed dispersers (G. Chen et al., 2017), and creators of products such as honey and wax. Biological control and pollination services are estimated to be worth several billion dollars in the United

States of America alone (Losey & Vaughan, 2006). Thus, maintaining these services is important for the future of agricultural production (Zhang et al. 2007).

2.2.2 Insect ecosystem services

Insect natural enemies are a group that generally attack pests of some kind. Many natural enemies have been identified, including predators, parasitoids, and pathogens (Landis, Wratten and Gurr, 2000). Biological control is the act of using natural enemies to control or manage pest organisms. A biological control agent can be successful at controlling a pest species if it either directly causes mortality of the pest (e.g. consumption or parasitism) or causes the pest to avoid a specific area (due to competition or risk avoidance) (Stenberg et al. 2021). Many groups of insects are biological control agents as they cause direct mortality of insect pests (Hawkins et al., 1997) or weedy plants (Kulkarni et al., 2015). Ground beetles (Coleoptera: Carabidae) are a diverse group of insects that have populated almost every habitat on earth. As such, they have a wide range of diets and adaptations (Lövei & Sunderland, 1996). Most carabids (73.5% of the 1054 species examined) are exclusively carnivorous, with smaller proportions being omnivorous (19.5%) or phytophagous (8%) (Laroche 1990). Many carabids will consume insect pests, molluscs, and seeds (Holliday et al. 2014), which helps to control pest populations. Carabids show potential for pest control in the laboratory, and field studies have shown reduced pest pressure of aphids, lepidopterans, slugs, and weeds (Kromp 1999). Carabids have commonly been used in entomological studies as they are numerous, rather easy to capture, and can be used as bioindicators for site health (Frampton et al. 1995; Kromp 1999). Due to this, and the well-defined taxonomic keys

from Lindroth for my study area of the prairie region of Canada (Lindroth, 1969a, 1969b, 1968, 1966, 1963, 1961), it was an easy decision to include the carabids as a focal species in my project.

Much like carabids, hover flies (Diptera: Syrphidae) are also predaceous—though primarily as larvae (Singh et al. 2020). While carabids generally prey on ground dwelling insect pests, syrphid larvae feed on soft-bodied insects that are higher in the canopy. Many species of syrphids specialize on aphids (Hemiptera: Aphidoidea) and each larva can consume up to 47 aphids per day (Curran 1920). Aphids themselves are crop pests (Ragsdale et al., 2004), so reducing their population size is a priority to agricultural production. Syrphids are somewhat unique in that they provide biological control services as larvae and pollination services as adults (Dunn et al. 2020). The adults are generalists and will visit a wide array of flowers (Klecka et al. 2018) including crops such as canola (Jauker & Wolters, 2008) and peppers (Jarlan et al., 1997), which may increase crop yield through increased pollination. Syrphid pollination efficiency is less than that of other pollinators, such as bees (Hymenoptera: Apoidea) (Jauker et al. 2012). Honey bees (*Apis mellifera* L.) have traditionally been used for commercial crop pollination services (Morse & Calderone, 2000), but their management has been increasingly difficult due to colony losses (Steinhauer et al., 2021). Wild bees that are already present within the landscape can be important pollinators of crops and may even improve yield above that of honey bees (Garibaldi et al., 2013; Greenleaf & Kremen, 2006). So, increasing reliance on wild pollinators may help to increase crop yield and lessen colony loss impacts.

The beneficial insects described above, particularly ground beetles and bees, contain a wide diversity of species with different life history traits (Laroche & Larivière, 2003; Michener, 2007). The diversity in species traits makes these groups an excellent choice to study how insect functional traits interact within the environment (see Laliberté and Legendre, 2010 for framework). Categorizing insects based on their life history traits (Bartomeus et al. 2018) may be more ecologically sound than categorizing them taxonomically, as not all members of a taxon may display the same traits. Similarly, insects may respond differently to habitat alterations based on their traits (such as activity timing, nesting locations, diet, etc.) (Caitano et al. 2020), but responses are inconsistent between landscapes (Bartomeus et al. 2018). Thus, understanding how insects in the prairie region of Canada interact with their landscape is important to determine best IPPM practices in this region.

2.2.3 Insect decline

Insects across the globe are in decline (Mancini et al., 2023; Sánchez-Bayo & Wyckhuys, 2019; Wagner, 2020; Wagner et al., 2021). Beneficial insects, such as pollinators, have been a focus of studies (Cameron et al., 2011; Graham et al., 2021; Potts et al., 2010) and declines have been observed for at least fifty years (Kevan 1975). Causes of insect decline are numerous and often compounded (Wagner et al. 2021) but a select few causes have been highlighted. Climate change and conversion of land to agriculture has been repeatedly identified as a major driver of population declines (Sánchez-Bayo & Wyckhuys, 2019; Wagner, 2020). Loss of dietary options (Wood et al. 2019), diseases (Evans & Schwarz, 2011), pesticides (Desneux et al., 2007), and a

combination of effects (Goulson et al. 2015) all work together to disturb insect communities. Many of these causes result in either population or species declines—a problematic trend as insect biodiversity increases ecosystem service sustainability (Kremen et al., 2002). Thus, understanding ways to support local insect populations, especially in agriculturally dominated landscapes, is important for conservation practices and the sustainability of insect ecosystem services.

2.2.4 Agricultural production

Prairie landcover once dominated the great plains of North America. Unfortunately, over 99% of historic prairies have been lost (Samson & Knopf, 1994) due to 200 years of land conversion (Henderson and Koper, 2014). Many of the insects that once relied on these areas are at risk of extinction (Hall, Catling and Lafontaine, 2011) due to this habitat loss and restricted resource availability. Meta-analyses often find that land conversion to agriculture is one of the main drivers of global insect declines (Wagner 2020). Rates of insect declines tend to increase with agricultural intensity (Mancini et al. 2023) and are observed even in protected areas (Hallmann et al. 2017). Agricultural production is a hostile environment for insects because it typically produces large fields with an emphasis placed on crop yield rather than maintaining ecosystem services (Landis 2017). Pesticide use, regular disturbances (e.g. tilling), and landscape simplification (e.g. monocultures) result in decreases in beneficial insect abundance or diversity (Potts et al. 2010) threatening the ecosystem services that these insects provide (van der Sluijs 2020). Recent agricultural policies have aimed to slow insect decline but,

unfortunately, these have proved insufficient (Mancini et al. 2023). Thus, additional interventions are necessary to maintain ecosystem services in agricultural systems.

2.3 INTEGRATED PEST AND POLLINATOR MANAGEMENT (IPPM)

2.3.1 Introduction to IPPM

Integrated Pest and Pollinator Management is an evolving framework that aims to support agricultural systems through pest control and pollinator management (Lundin et al. 2021). This concept has evolved from past frameworks including Integrated Pest Management (IPM) (Barzman et al., 2015; Kogan, 1998) and more recently, Integrated Crop Pollination (ICP) (Isaacs et al. 2017), which focuses on pollinator management. At its core, IPPM relies on proactive and biodiversity-based approaches (Lundin et al. 2021) with the goal of designing agricultural systems that are balanced enough that minimal intervention is necessary to control pest outbreaks while maintaining other beneficial insects, like pollinators (Ratnadass & Barzman, 2014). The key here is to use proactive, rather than reactive, measures to manage insects. Proactive measures include landscape and crop management, which can alter insect assemblages (Gurr et al. 2017) before an outbreak occurs.

2.3.2 Landscape management for IPPM

The foundation of IPPM practices occurs through habitat and landscape management (Lundin et al. 2021). The amount of semi-natural habitats in the

surrounding landscape can support natural enemies and their associated pest control (Rusch et al. 2016) as well as pollinators (Kennedy *et al.*, 2013) and their ecosystem services. Additionally, increasing crop diversity can help to limit pest outbreaks (Delaune et al., 2019; Root, 1973) and increase natural biological control (Redlich et al., 2018). Landscapes with irregularly shaped fields and patches of natural habitats benefit both pest suppression and pollination (Martin *et al.*, 2019). This gives general guidelines for a model landscape, but in practice, each region is unique and creating natural areas is no easy task. Thus, understanding how insects respond to their given landscape (Karp et al. 2018) and how to support local populations is a crucial step in IPPM practices.

The landscape in IPPM studies encompasses the immediate area surrounding an organism, but the exact extent of this range depends on the study species. With insect studies, especially pollinators, we often study the landscape within buffers of a 3 kilometre (km) radius of the sites as the flight range of these insects often falls within this window (Greenleaf et al. 2007). A range of buffers should be used in insect studies as insects experience the landscape in different ways. Karp et al. (2018) found that landscape composition explained insect responses within individual studies, but responses were inconsistent across studies. Thus, the way that insects interact with the landscape is often dependent on local factors and changes between regions. There are several hypotheses for how insects interact with the landscape provided by Tschamntke et al. (2012) and I will focus on two of them here: the intermediate landscape complexity hypothesis and the cross-habitat spillover hypothesis. Both hypotheses try to explain how insects move within and experience the landscape.

The intermediate landscape complexity hypothesis predicts that insect management practices will be more effective in landscapes with intermediate levels of complexity, rather than simple or complex landscapes. Simple landscapes lack the insect diversity needed to respond to management practices and complex systems accrue limited benefit from additional complexity. The cross-habitat spillover hypothesis predicts that the movement of organisms across habitats has functional consequences on energy, resources, and functional traits within the landscape. Specifically, the spillover of beneficial insects into crop fields has functional consequences if pollination or biological control are increased, as these both directly benefit the crop (Griffiths et al., 2008; Holzschuh et al., 2012; Obrycki et al., 2009). Landscape composition (Kennedy et al. 2013), configuration (Olynyk, Westwood and Koper, 2021), and local traits (Rowe et al. 2021) all influence insect assemblages and the strength of spillover effects (Kemmerling et al. 2022). Thus, the landscape may predict how readily insects move between patches (Riggi et al. 2024) and these can have functional effects on the landscape (Blitzer et al. 2012). Thus, understanding landscape traits that influence beneficial insect assemblages in my study region is important for effective IPPM.

2.3.3 Composition and configuration

Researchers often break landscape variables into two main categories: composition and configuration. The composition of the landscape refers to the overall makeup of landcover types, generally referring to the total percentage of each land category within a specific area. Insects often respond positively to natural and diversified

landcover (Kennedy et al. 2013). This is likely because these landcover types provide sufficient resources for insects.

Configuration refers to the position of patches relative to each other and is generally used to determine levels of habitat fragmentation, complexity, or diversity. As Landis et al. (2000) points out, the relative position of these landcover types within the landscape must be spatially and temporally favorable for the insects. Insects move between landcover types for different resources (Riggi et al. 2024) and this movement may influence their ecosystem services (Samaranayake & Costamagna, 2019). Insects often respond to landscape connectivity (Rösch et al. 2013) as this may increase the likelihood of intercepting necessary resources. Agriculture can decrease landscape diversity through the conversion of existing land into monocultures, so finding ways to increase landscape complexity in these areas may better support local insect populations (Estrada-Carmona et al. 2022).

2.4 LANDSCAPE ENHANCEMENTS FOR BENEFICIAL INSECTS

2.4.1 Introduction to landscape enhancements for beneficial insects

Alterations to the landscape can be made to increase beneficial insect abundance, ecosystem services, or conservation efforts. In agricultural landscapes, hedgerows, beetle banks, or floral enhancements are often used to support beneficial insects. These habitat manipulations can increase beneficial insect (Landis et al., 2000) and / or decrease pest (González-Chang et al. 2019) abundance and diversity. Pest populations could be altered

independent of natural enemies. Trap cropping is the use of an attractive habitat manipulation to attract insect pests to a single location– where they can then be controlled across a smaller spatial scale (González-Chang *et al.*, 2019).

Increases in local vegetation diversity through habitat enhancements can increase insect diversity within the enhancement and thus, increase landscape level insect diversity as well (Gurr *et al.* 2017). Habitat modification efficiency depends on the local landscape, as enhancements placed in diverse, rather than simple, landscapes tend to show stronger effects (Bianchi *et al.*, 2006). Complex agricultural landscapes overall support more biodiversity than simple landscapes (Estrada-Carmona *et al.* 2022) so finding ways to increase habitat diversity is essential for sustainable conservation practices. A review of common landscape enhancements to support beneficial insects is below.

2.4.2 Hedgerows

Hedgerows are strips of (generally) woody vegetation generally located between fields. These originated in agriculture and have been used as a wind break, to minimize water and soil erosion, and to separate fields (Burel 1996). Insects benefit from hedgerows as they provide nutrients, refuge, and a corridor to aid in movement (Alignier *et al.*, 2020). Hedgerows containing tree and shrub species can slow wind speeds up to 16 times their height, creating a different microclimate from that in the surrounding area (Forman & Baudry, 1984). This microclimate can cause increases in plant diversity, ultimately increasing diversity of food and shelter available to insects (Maudsley 2000). Many insects will use blooming fruit trees and shrubs commonly found in hedgerows for

nutritional rewards (Gareau et al., 2013; Jacobs et al., 2009). Hedgerows not only support insects on field edges but can increase insect abundance and ecosystem services within adjacent fields (Morandin et al., 2014), which can decrease pest populations (Girma et al., 2000). Overall effects depend on the composition of the hedgerow, with sown perennial species being more attractive than volunteer weedy species (Gareau et al., 2013).

2.4.3 Beetle banks

Beetle banks are strips of vegetation that occur either on field edges or within fields. Traditionally a mound of soil is created with a plow and then various grass species are sown onto the mound (see MacLeod et al., 2004). This provides areas where beetles and other organisms can overwinter and reproduce without disturbance (see Cardarelli and Bogliani, 2014 regarding disturbance intensity). Beetle banks have been found to increase ground-dwelling insect densities more than hedgerows, but population fluctuations were more variable within the banks (Collins et al. 2003). Research generally finds increased beetle density and diversity (Thomas, 2001) and increased pest suppression in adjacent fields (Collins et al. 2002) when using beetle banks. A long-term study found that vegetation within the banks was relatively simple compared to the surrounding areas initially. After four years, beetle density was higher within the banks and beetle diversity increased over the seven years of the study (MacLeod et al. 2004).

The main distinction between beetle banks and floral enhancements is the inclusion of flowering forb species in the latter. Most research on traditional beetle banks was published before roughly 2005 with more research on floral enhancements picking

up around the same time. Comparisons of flowering and grass dominated mixtures found that beetle density and diversity were higher within the flowering mixtures (Woodcock et al. 2005). Thus, researchers focused more on the inclusion of flowering species within habitat enhancements to support a wider array of insect species.

2.4.4 Floral enhancements

Floral enhancements, pollinator strips, wildflower strips, and floral strips, are some of the names used to describe sown floral species that are often placed near agricultural fields to increase beneficial insect abundance and diversity. Generally, a mixture of floral and grass species is planted near the crops, which provides resources and shelter for beneficial insects both within and between years. These enhancements can either be in block form (Blaauw and Isaacs, 2014), in strips along the field edges (Kordbacheh et al., 2020), or in other shapes near the crop (Sutton et al. 2017). Studies on floral enhancements have been conducted world-wide, but studies from North America are mostly limited to perennial crops (Zamorano et al. 2020). Crops are consistent between years in perennial systems, while in rotationally managed agriculture the crops often change. Heterogeneity in the landscape can support more beneficial insects (Estrada-Carmona et al. 2022) but large field sizes make finding resources difficult for insects. Floral enhancements provide a safe space where insects have consistent access to food and shelter between years and may help to sustain insect populations in years with low resource availability.

Floral enhancements take many shapes and sizes, but some general best practices have been established. At least 3–5% of the arable landscape should be uncropped and

managed for biodiversity with specific goals (Sutton et al. 2017). These goals could be to provide forage (see Filipiak, 2019 for pollen nutrition and insect development), provide overwintering areas (Graham et al. 2020), or other goals. Enhancements may be created for conservation of specific species or a general insect population and these goals may influence floral selection.

2.4.5 Floral selection

The mixture of floral species is often region dependant and should be selected with purpose (Hatt et al. 2020) as not all flowers are equally as attractive to beneficial insects (Carrié et al., 2012). Flowering time, establishment success, and known insect affiliations should drive seed selection, though seed availability and cost should also be considered. If floral enhancements flower during adjacent crop bloom, then they may compete with the crop for pollinators. Because of this, some studies choose to only include floral species that bloom before or after the adjacent crop (Blaauw & Isaacs, 2014, 2015). Floral species should complement each other and should be selected so that there are blooming resources within the strip across the entire season. Establishment success is often a hurdle with floral enhancements, as differences in local soil and weather conditions may make establishment difficult. Establishment depends on preparation of the seed bed, seeding method, and post seeding care such as irrigation and weed control (see Aldrich and Dumroese, 2002 for general recommendations).

Plant species should be selected based on the target insects as some plant traits may attract different insect groups (Hatt et al. 2019). Overall floral area is highly attractive to both pollinators and other beneficial insects, so selecting species with large

or numerous flowers may increase effects of floral enhancements (Rowe et al. 2021). Additionally, Rowe et al (2021) found that natural enemies were attracted to floral area and overall pollen abundance, so selection of species with high pollen production should benefit insects. Some plant species have been found to attract a generalist group of beneficial insects as *Achillea millefolium* and *Solidago* species are often highly attractive (Gibson et al. 2019). Thus, these species should be considered as part of the seed mix if possible.

2.4.6 Insect responses to floral enhancements

Floral enhancements tend to increase abundances of beneficial insects but exact responses often vary between studies. Review studies (Albrecht et al., 2021; Zamorano et al., 2020) found that floral enhancements increased beneficial insect capture rates on field edges over control comparisons. However, whether these insects spill over into the adjacent crops was inconsistent. There are two leading hypotheses for how beneficial insects move in relation to floral enhancements- spillover (Morandin & Kremen, 2013) and concentration (Kleijn et al. 2018). The difference between these two hypotheses is whether the beneficial insects are moving from the enhancement into the landscape or vice versa. Studies that do identify a spillover effect note that insect abundance decreases as distance from the field edge increases (Albrecht et al. 2021) and that this effect may only reach 100 m into the fields (Morandin et al., 2014). Many studies find that it may take 3–4 years before differences in insect communities within enhancements are detected (Blaauw & Isaacs, 2014; MacLeod et al., 2004). Given that many studies only include the first three years of data (Zamorano et al. 2020), it could be possible that floral

enhancements concentrate insects initially and spillover effects only occur several years after establishment. Older plantings that contain a diversity of perennial species are more effective at enhancing pollination services (Albrecht et al. 2021). Thus, the need for long term studies on floral enhancements as vegetation communities establish is necessary to effectively understand and use this management tool.

The underlying mechanisms behind beneficial insect responses to floral enhancements are not well studied. Beneficial insects require food, shelter, and access to mates to effectively reproduce. Floral enhancements have been shown to increase nesting rates and reproductive success and lower parasitism rates of bees (Ganser et al., 2021) which may increase population size over time. These enhancements generally increase the floral resource available to beneficial insects, but whether beneficial insects utilize this pollen directly varies between studies. Graham et al. (2020) found that most of the pollen collected was from non-sown species while Ganser et al. (2021) found the opposite. So responses may be region and landscape dependant. Few studies analyze reproductive output in relation to floral strips and none have studied their effects in rotational systems with naturally occurring beneficial insect populations. Understanding the mechanisms that support beneficial insect populations allows researchers to apply findings on landscape management more broadly than studying the effects of floral enhancements alone.

2.5 CROP MANAGEMENT FOR IPPM

2.5.1 Introduction to crop management for IPPM

Another proactive measure that is taken in IPPM practices revolves around how farm management practices may impact insect populations. These have collectively been termed cultural control methods (Bajwa & Kogan, 2004). Overall, organic farming may reduce pest pressure when compared to conventional systems (Török et al. 2021). Thus, the combined actions of organic agricultural production may better support insects and their ecosystem services than conventional farming (Morandin & Winston, 2005). Practices to support insects include increasing crop and floral diversity and altering current practices such as tillage, seeding date, crop rotation, and others (Lundin et al. 2021). Crop diversity can be increased through intercropping (Iverson et al. 2014) or cover cropping (Schipanski et al. 2014), which generally results in increases in pollinator abundance (Mallinger et al. 2019). Vegetation diversity can be increased through the addition of sown floral enhancements or by reducing weed control to promote natural vegetation (DiTommaso et al. 2016). Tilling soil is a common weed control technique, but conventional tillage provides intensive and widespread disturbances that result in lower beneficial insect population sizes (Alyokhin et al., 2020).

2.5.2 Grazing, haying, and vegetation removal

Livestock grazing (Teague & Kreuter, 2020) may improve soil and insect health. Grazing provides less intensive vegetation removal than haying, with time between removals, for plants and insects to recover— though overall effects vary based on grazing intensity and timing. Studies generally find that high grazing intensity negatively effects beneficial insect abundance and diversity (Mayer, 2004) while moderate intensity may

result in a net benefit. Grazing later in the season may result in more pollinators than early grazing (Sjödin, 2007). Including grazing into agriculturally intense areas helps to increase landscape heterogeneity and increase beneficial insect abundances (Morandin et al., 2007). However, pastureland in the Morandin et al. study was close (< 800 m) to fields, so it is uncertain how grazing affects insects at broader landscape scales. Overall effects of grazing may be more linked to the plant species present within the sites (Yoshihara et al., 2008). If livestock are not allowed to graze, then fields are hayed for feed instead. The act of haying fields results in a mass removal of vegetation, but leaving strips uncut may provide long term benefits to insects (Buri, Humbert and Arlettaz, 2014). Grazing, haying, and other techniques have been used to manage grasslands (P. B. Martin, 1983) but how these management techniques influence insect assemblages in Manitoba is unclear.

2.6 RATIONALE OF THIS STUDY

The goal of my study was to understand responses of beneficial insect groups to IPPM practices that could be implemented in annually managed agricultural areas within the prairie region of Canada. Particularly, I investigated the effects of floral strips adjacent to annually managed crop fields on beneficial insect communities. I hypothesized that the addition of floral strips would increase insect capture rates and diversity compared to control and natural sites. Additionally, I hypothesize that beneficial insect capture rates will be higher within the field adjacent to the floral strip than the field adjacent to a grassy margin. I also tested the relative effects of grazing and hay practices on beneficial insects. I tested for differences in beneficial insect abundance, diversity,

community structure, functional traits, and reproductive output between treatments in these studies. I hypothesized that grazed areas will capture a unique community of beneficial insects compared to the other two treatments (hayed and natural) and that insect capture rates will vary based on functional traits. I compared all studies to semi-natural areas to determine differences in insect communities between agricultural and non-agricultural habitat types. The landscape was investigated to determine relationships between insects, management techniques, and the surrounding environment.

CHAPTER 3. FLORAL STRIPS ADJACENT TO MANITOBA CROP FIELDS ATTRACT BENEFICIAL INSECTS SHORTLY AFTER ESTABLISHMENT REGARDLESS OF MANAGEMENT TYPE OR LANDSCAPE CONTEXT

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3.1 ABSTRACT

1. Insects provide valuable ecosystem services to agriculture, but their populations are often lower in these areas.
2. We established floral strips on the edges of eight organic and seven conventional crop fields in southern Manitoba. We then compared the capture rates of ground beetles, syrphid flies, and bees in floral strips, control (unmodified) field edges, nearby semi-natural sites, and within the strip and control fields at six distances away from field borders (5–150 m). The effects of farm management and landscape context on insect abundance, diversity, and associated species traits were investigated.
3. Capture rates of bees and beetles were significantly higher within the floral strips than the control treatment. However, capture rates of syrphids were significantly higher in the control. Beetles and bees responded positively to blooming forb cover and edge density (0.5 km) while syrphids responded positively to nonrewarding agriculture and negatively to the semi-natural areas (1 km). We found no significant difference in insect abundance or diversity between organic and conventional management types and no differences in insect abundances within the fields were detected. No significant associations between insect traits and landscape variables were identified.

4. These results show that floral strips are effective at attracting ground beetles and bees shortly after establishment, but these communities are functionally similar to those present in control fields. Longitudinal studies are needed to determine if populations continue to increase over time and become more functionally diverse in fields with floral strips.

Keywords

Functional Traits, Natural Enemies, Overwintering Habitat, Planting, Pollination, Wildflower Strip

3.2 INTRODUCTION

Insects provide critical ecological functions in terrestrial ecosystems. Ecological functions with direct benefits to humanity have been termed ‘ecosystem services’ (Daily et al. 2003) and have been estimated to be worth roughly \$57 billion in the United States alone as of 2006 (Losey & Vaughan, 2006). Predation and pollination are important regulating ecosystem services to agriculture. Predators control populations of herbivorous insects (Hawkins et al., 1997). For example, ground beetles (Coleoptera: Carabidae) and syrphid larvae (Diptera: Syrphidae) provide biological control of pest species (Gaines & Gratton, 2010; Kaiser et al., 2007). Pollinators, such as bees (Hymenoptera: Apoidea) and adult syrphids, support plant reproduction, ultimately increasing primary production (Woodcock et al. 2019). Ecosystem services in agriculture increase yield and ensure sustainable food production (Durán et al. 2010). However, agriculture has often been associated with the declines of insects that support ecosystem

services (Grixti et al., 2009; Sánchez-Bayo & Wyckhuys, 2019). Thus, agronomic practices that conserve beneficial insects should be a priority (Gayer et al. 2019).

Insect declines have been detected around the globe (Cameron et al., 2011; Goulson et al., 2015; Potts et al., 2010; Zattara & Aizen, 2019) highlighting agriculture and habitat loss as major drivers of these declines (Hallmann et al., 2017; Wagner, 2020; Whitehorn et al., 2012). Management practices that support both agricultural production and insect diversity are needed for long term sustainability (Seibold et al. 2019). Agriculture in Manitoba is intensively managed and field sizes are typically large, with an average crop area per farm of > 1,000 acres (Census of Agriculture historical data 2021). In this area, farming is the predominant landcover type, with over 47% of the available land in Manitoba south of the 51st parallel used for agriculture (Agriculture and Agri-Food Canada, 2021). Floral strips installed next to crop fields have proven effective at increasing beneficial insects in some systems (Blaauw and Isaacs, 2015, 2014; Haaland et al., 2011; Kordbacheh et al., 2020; Morandin et al., 2014; Williams et al., 2015) because they provide insects with food and refuges (Haaland et al., 2011; Jervis et al., 1993; Langellotto & Denno, 2004). Floral strips can increase predators and decrease pests up to 200 metres into crop fields (Morandin et al., 2014), an effect known as “spillover” (Blitzer *et al.*, 2012). Other studies show that floral habitats increased pollinator abundances on farms and were also associated with greater crop yield three to four years after establishment (Blaauw and Isaacs, 2014). Other insects, such as ground beetles, benefit from the structured habitat (Langellotto & Denno, 2004) and overwintering areas (MacLeod et al. 2004) that floral strips provide. Strips used for beetle conservation are often called ‘beetle banks’ and are common in Europe (MacLeod et al. 2004). These

have been associated with increased predator and decreased pest populations that spillover from the strip into the field (Collins et al. 2002), increasing ecosystem services within the crop (Morandin et al., 2014). Floral strips have also been shown to increase the diversity of beneficial insects (Kordbacheh et al., 2020; N. M. Williams et al., 2015). Increases in diversity can support ecosystem stability (Hooper et al. 2005) and even improve crop yield (Garibaldi et al. 2016; Woodcock et al. 2019). It is generally understood that floral strips increase beneficial insect abundances near relatively consistent, perennial, crops (Blaauw and Isaacs, 2015, 2014; MacInnis et al., 2020). However, the effects of floral strips in areas with large scale rotational cropping systems have not been adequately studied. If floral strips prove effective in these systems, then they may help to offset some insect populations declines and help increase ecosystem services to agriculture.

Pest species can cause drastic losses to crop yield (Ragsdale et al., 2004, 2007) resulting in a 25% loss of yield or higher in some cases (Beckendorf et al., 2008). Agricultural pests including some insects (Beckendorf et al., 2008), weeds (Swinton et al. 1994), and diseases (Strelkov & Hwang, 2014) can lower crop yield. Ground beetles have been found to decrease weed seed abundance by up to 55% (Ward et al. 2011). Some species of ground beetles also show potential for management in agriculture for weed control (Kulkarni et al., 2015) given that many species are associated with cultivated lands (Holliday et al. 2014). Ground beetles show great variation in species traits such as mobility, diet, and activity (Larochelle & Larivière, 2003), meaning that they likely fill multiple functional roles in the environment. Syrphids provide both biological control and pollination services in their different life stages. As larvae,

syrphids are efficient predators of aphids and have been used for biological control in agriculture (Bugg et al., 2008; Curran, 1920; Hopper et al., 2011). As adults, syrphids are pollinators of plants such as mustards (Singh et al. 2020), strawberries (Sharma et al. 2014), and many wildflowers (Tooker et al., 2006). Wild bees are also important pollinators of many crops such as alfalfa (Pitts-Singer and Cane, 2011), blueberries (Blaauw and Isaacs, 2014; Isaacs and Kirk, 2010), cherries (Holzschuh et al., 2012), and sunflowers (Greenleaf & Kremen, 2006). Wild bee pollination results in more pollen deposition per visit of some crops (Javorek et al., 2002; Park et al., 2016) which consequently increases crop yield (Garibaldi et al. 2013) and quality (Klatt et al. 2013). Since wild bees and syrphid flies are important pollinators of many crops (Dunn et al., 2020; Rader et al., 2016) and ground beetles are important predators of pest species (Laroche & Larivière, 2003), increasing their abundance should be of importance to growers.

The landscape that surrounds floral strips may influence their effect on insect community assemblages (Tscharntke et al. 2005) and their associated traits such as body size (Warzecha et al. 2016) and pollination ability (Hoehn et al. 2008). The amount of high-quality habitat around agricultural fields is positively associated with many insect groups including ground beetles, syrphids, and bees (Kennedy et al., 2013; Purtauf et al., 2005; Wratten et al., 2003). These areas can also act as a source of beneficial insects that may then spill over into the adjacent crop field (Montero-Castaño et al., 2016). Unfortunately for insects, the prairie provinces are dominated by intensive agriculture, often leaving few semi-natural areas in the landscape (Agriculture and Agri-Food Canada, 2021). Since floral strips may be particularly

important in landscapes with limited floral resources (Scheper et al. 2015), installing floral strips in the prairie provinces may greatly benefit insects. Farm management can also influence insect abundance (Purtauf et al. 2005). Insecticides may result in direct mortality of beneficial insects (Whitehorn et al. 2012), but worldwide, herbicides are applied most frequently (Donaldson et al., 2002). Organically managed fields tend to use less insecticides and herbicides that in turn sustain higher diversity of floral resources and larger populations of pollinators than conventionally managed sites (Kennedy et al. 2013; Lichtenberg et al. 2017). But these responses are inconsistent and can even vary among bee species in the same genus (Rundlöf et al., 2008). Thus, it is important to identify which insect species or functional traits are supported most by floral strips, as different functional groups may not respond in the same way (Bartomeus *et al.*, 2018). Assessing floral strips in different management and landscape contexts in Manitoba is important for understanding how to effectively use this strategy to increase abundances of beneficial insects in this region.

At multiple farms across southern Manitoba, we set out to answer the following questions: 1) How do beneficial insect communities (ground beetles, syrphid flies, and wild bees) respond to establishment of floral strips? 2) Do beneficial insects spillover into crop fields? 3) How do farm management (organic or conventional) and landscape context affect beneficial insect communities? 4) Do functional traits of beneficial insects predict their responses to floral strips?

3.3 METHODS

3.3.1 Site selection, description, and experimental design

Sites were selected using multiple criteria for inclusion in the study. A call for volunteers was placed and farmers then offered their land for participation in the study. For a farmer to be included, they must be located within the agricultural region in Manitoba and provide at least two sites located within a 10 kilometre (km) radius that are managed in the same way and growing the same crops. One of these two sites was selected by the farmers to establish a floral strip and the other was used as a control. At the floral strip fields, one edge was selected by the farmers to be enhanced with a floral mixture (strip treatment) and a separate field edge at least 150 m away from the floral strip was left unmanaged as an on-site control (on-site control treatment). At the separate control fields, we selected one unmanaged edge to sample (control treatment). The control field was used as a spatially independent site, whereas the on-site control allowed for comparisons with more comparable land use conditions, but non-independent insect communities. Within 10 km of the floral strip and control sites, we selected a semi-natural site (semi-natural treatment) to use for comparisons. Semi-natural sites were selected by using aerial imagery (Google Earth, 2001) and information obtained by farmers to identify any possible unmanaged areas in the landscape. We verified semi-natural areas were unmanaged before sampling by observing the land for signs of human management within the last five years. Often, semi-natural sites were riparian areas since other unmanaged areas within 10 km of sites were sometimes unavailable or inaccessible. This treatment was used to determine if floral strip communities were similar to unmanaged sites in the landscape.

Of the sites volunteered for inclusion in the study, fifteen fields were selected to establish floral strips. These same fifteen sites each had an on-site control selected on an unmanaged edge. Eleven separate sites were selected as control fields, and fourteen semi-natural sites were selected for sampling. In total, 55 sites were selected for sampling. Sites were generally located in a cluster that included one floral strip, on-site control, control, and semi-natural within a 10 km radius. We identified 15 site clusters for this study. In some cases ($n=4$) clusters did not contain all treatments if a control or semi-natural site was not available within the 10 km radius.

Average field size (strip and control treatments) was 58.3 ± 10.73 (mean $\pm$ SE) hectares and field width varied from 300–800 m. Eight of these field clusters were under organic management (certified organic by the Canadian Organic Standards) and seven clusters were under conventional management (Figure 3.1). Crops grown within a specific site cluster were the same. The crops, chosen by the farmers, that were grown in each site cluster included oat and pea intercrop ($n=4$), canola ($n=2$), oat ($n=2$), soybean ($n=2$), wheat ($n=2$), corn ($n=1$), flax ($n=1$), and forage ($n=1$).

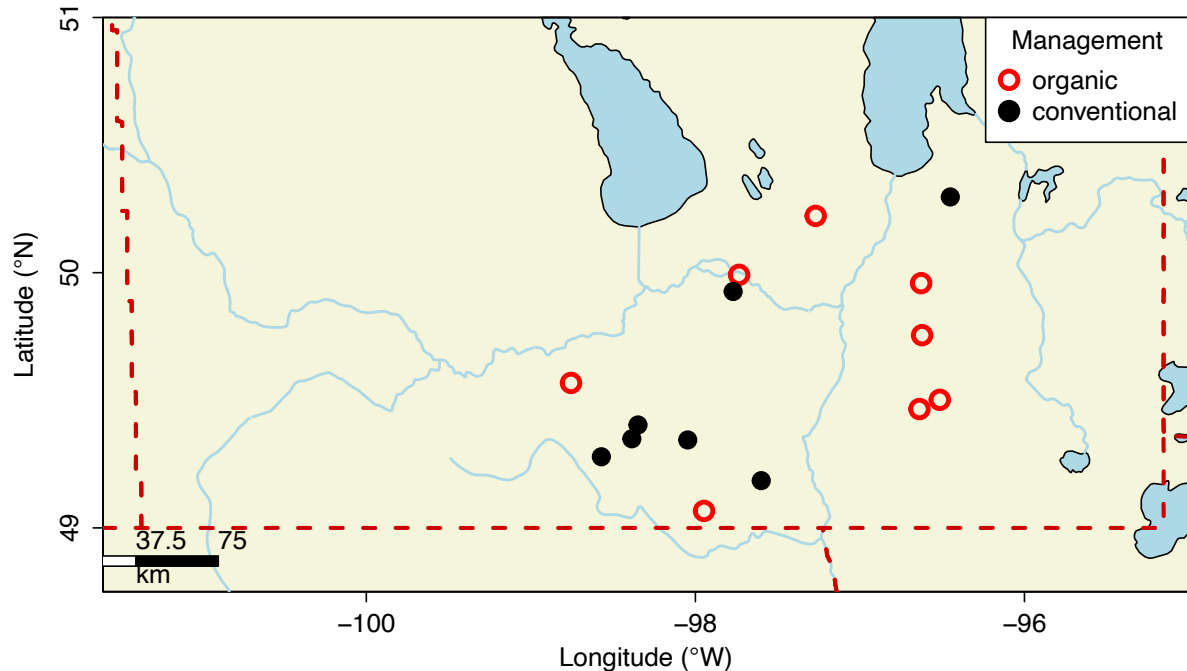


Figure 3.1 Locations of all blocks of sites where beneficial insects were sampled at farms in southern Manitoba during the summer of 2019. All sites were sampled once for each collection method. Dashed red lines represent borders between Manitoba, Ontario, North Dakota, and Minnesota. This figure was created using the PBSmapping package in R (R Core Team, 2023; Schnute et al., 2021). Sample sizes: conventional (allows for the use of insecticides, herbicides, and fungicides), $n=7$; organic (certified organic by the Canadian Organic Standards), $n=8$.

3.3.2 Floral strip establishment

The area selected for floral strip establishment was prepared by tilling or disking to control existing vegetation and prepare the seedbed. Floral strip size was 0.64 ± 0.097 (mean $\pm$ SE) ha (range 1.58–0.15 ha) and was 8.0 ± 0.75 metres wide (range 3.65–13.4 m). The seed mix for the floral strips was broadcast at a rate of 22.4 kg/ha and lightly harrowed into the soil between May 30 and June 6, 2019. The seed mix contained 15 species of forbs and four species of grasses (Table S 7.1). The most common seeded species were *Fagopyrum esculentum* (Moench) (34.4%), *Onobrychis vicifolia* (Scop.) (26.5%), *Thinopyrum intermedium* (Barkworth & D.R. Dewey)

(11.2%), and *Helianthus annuus* (L.) (5.6%). Floral species were chosen based on their availability and their ability to provide ample cover in early years (to prevent weed establishment) and to provide insects with a range of flowers that bloom over the entire season.

3.3.3 Pitfall traps

Insect sampling took place once plants in the floral strips started to bloom in mid-July. Pitfall traps were used to collect ground beetles in the field margins and floral strips. Five pitfall traps, spaced 5 m apart, were deployed in each of the 55 sites. To test if beneficial insects spill over from the field edges into the crops, we placed one pitfall trap at 5, 15, 25, 50, 100, and 150 m away from the floral strip and the grassy border of the control field. Pitfall traps were left in the fields for seven days. Samples were collected between 18–30 July 2019 and was conducted once per site. Each pitfall trap consisted of two plastic 1 L deli cups nested within each other. Traps were placed so the lip of the cup was at ground level and were filled with roughly 250 ml of 50% propylene glycol and water mixture during the collection period. A 15x15 cm piece of wire mesh with 2.5x2.5 cm holes was placed on top of the deli cups to exclude larger animals. Sampling took place during clear weather, but a plastic cover was suspended above the trap in case of unexpected rains.

3.3.4 Pan traps

Pan trap stations were used to collect bees and syrphid flies in the same locations as the pitfall traps (five pan trap stations in each of the field edges and one pan trap at each of the six distances within the fields). Each station included three 82 ml pan traps (one yellow, blue, and white) on a wooden platform suspended at the height of the surrounding vegetation by a metal pole. Pan traps were deployed in the morning and removed each day after other sampling methods were completed. Sampling took place between 31 July-22 August 2019 and was conducted once per site. Sampling hours for pan traps was recorded and used as an offset in generalized linear models. Average length of sampling periods was 4.95 ± 0.18 (mean $\pm$ SE) hrs.

3.3.5 Targeted pollinator netting

Bees were netted for 15 minutes along a 30 m transect in each of the floral strips, on-site controls, controls, and semi-natural sites. One net transect per treatment was used. The timer was stopped when bees were being removed from the net. Pollinators were separated based on host plant and brought to the lab for further processing. Netting coincided with pan-trapping between 31 July-22 August 2019 and was conducted once per site.

3.3.6 Specimen processing

Ground beetles, bees, and syrphids were pinned and identified to species or morphospecies using taxonomic literature, reference to expertly identified specimens,

and checked by taxonomists when possible. Ground beetles were identified largely with keys by Lindroth and Bousquet (Bousquet, 2010; Lindroth, 1961, 1963, 1966, 1968, 1969a, 1969b). Syrphids were identified using Skevington et al. (Skevington et al. 2019). Bees were identified to genus (Packer et al., 2007) and then species using appropriate keys for each family: Andrenidae (Bouseman & LaBerge, 1979; LaBerge, 1967, 1969, 1971, 1973, 1977, 1980, 1986, 1987, 1989; LaBerge & Bouseman, 1970; LaBerge & Ribble, 1972, 1975; LaBerge & Thorp, 2005; Ribble, 1967, 1968, 1974; Shinn, 1967), Apidae (Droege et al., 2010; Droege & Rehan, 2016; LaBerge, 1956b, 1956a, 1961; Laverty & Harder, 1988; Onuferko, 2017, 2018; Rehan & Sheffield, 2011; P. H. Williams et al., 2014), Colletidae (Snelling, 1966, 1968, 1970, 1983; Stephen, 1954), Halictidae (Coelho, 2004; Gibbs, 2010, 2011; Gibbs et al., 2013; McGinley, 1986, 2003; Sandhouse, 1941), and Megachilidae (Baker, 1975; Gonzalez & Griswold, 2013; Michener, 1947; Sandhouse, 1939; Semmler et al., 2017; Sheffield et al., 2011).

3.3.7 Floral surveys and weather data

Ground cover was measured using a 1 x 0.5 m quadrat placed every five metres along each of the net transects described in section 1.4. Flowers were identified to genus or species (Newcomb, 1989; Scoggan, 1957) which was then used to describe floral diversity within each transect. The percent cover of each blooming forb species, and total cover of grasses, forbs (blooming and non-blooming), and bare ground was visually estimated for each quadrat using the figures provided by Folk (1951). At each sample, cloud cover (sunny, partly cloudy, or overcast) was determined visually.

Average and maximum wind speed and ambient air temperature were recorded with a Kestrel 3000 pocket weather meter (Philadelphia, PA).

3.3.8 Landscape analysis

Land use data was extracted from the 2019 annual crop inventory from Agriculture and Agri-Food Canada (Agriculture and Agri-Food Canada, 2021) using the *landscapemetrics*, *raster*, and *sampSurf*, packages in R (Gove, Jeffrey, 2021; Hesselbarth et al., 2019; Hijmans, 2021; R Core Team, 2023). We separated crops into rewarding and non-rewarding categories since insects respond differently to flowering versus non-flowering crops (Holzschuh et al., 2013; Le Féon et al., 2013; Montero-Castaño et al., 2016). The rewarding agriculture category included crops with flowers frequently visited by insects for floral rewards, such as canola, flax, soybeans, peas and beans, and other vegetables. Nonrewarding agriculture included nonflowering crops such as cereals. The semi-natural category included land use types such as shrubland, forest, and wetlands. The urban category included permanent human made structures such as roads, buildings, and towns. We extracted landscape composition data as the percent of semi-natural, rewarding agriculture, nonrewarding agriculture, and urban areas at six spatial scales (0.5, 1, 1.5, 2, 2.5, and 3 km) using the *lsm_c_ca* function from the *landscapemetrics* package in R. Landscape data was extracted at multiple spatial scales because the different insect taxa will likely respond to different spatial scales (Karp et al. 2018). We then extracted landscape configuration data as the edge density (sum of the edges of each individual patch type, including field boundaries, in relation to landscape area) and landscape Shannon diversity using the

lsm_l_ed and the *lsm_l_shdi* functions, respectively from the *landscapemetrics* package in R (Hesselbarth et al. 2019; R Core Team 2023). These metrics were also extracted for each of the six spatial scales described above.

3.3.9 Data analysis

We examined compositional differences in bee communities among treatments and management styles using distance-based redundancy analysis using the *capscale* function and permutational multivariate analysis of variance (PERMANOVA) using the *adonis2* function from the *vegan* package (Oksanen et al. 2019). Treatments included strip (edges with a floral enhancement), control (edges of crop fields that are unmanaged, including the on-site control and control field), and semi-natural (semi-natural sites). We tested for differences between the on-site control and the control samples for each collection method and found no significant differences. We also ran analysis with the on-site control samples removed and found no differences, so on-site controls and control field margins were grouped together for all analyses since they are both unmanaged field edges of crop fields. Management types included conventional (use of insecticides, herbicides, and fungicides) and organic (certified organic by the Canadian Organic Standards). Ordinations were made using the *capscale* function and a Bray-Curtis dissimilarity measure. These were then tested with the *adonis2* function using the FDR corrected p-value from the *vegan* package in R (Oksanen et al. 2019; R Core Team 2023). Permutations were restricted by blocks of sites using the *permutations* argument in *adonis2*. If the results of the PERMANOVA were significant, then the *pairwise.adonis* function was used from the *pairwiseAdonis*

package (Martinez Arbizu 2017) to test for differences among treatments. Significant effects in PERMANOVAs may be due to differences in group centroids or group dispersions (or both) (Anderson 2017), so we examined group dispersions separately using the *betadisper* and *permutest* functions in the vegan package (Oksanen et al. 2019) and using the *TukeyHSD* function (R Core Team 2023). The number of permutations for permutational based tests was 999. Seeds were set before permutational analysis for reproducibility.

Separate generalized linear mixed effect models (GLMM) from the glmmADMB and glmmTMB packages were created in R to test for differences in floral composition, insect abundance, and insect diversity among treatment groups or management styles as well as the interaction between treatment and management. (Brooks et al., 2017; Fournier et al., 2012; R Core Team, 2023). We also created separate models to test for differences in insect abundance or diversity as distance from the field edge (floral strips or grassy margins of control fields) increased. Site was used as a random factor for all models. We first checked that the best error structure was applied to each model by comparing Akaike Information Criterion (AIC) scores, log likelihood, and examining qq and heteroscedasticity plots. For percentage (vegetation) data, beta error-distributions were used. The beta distribution is restricted to values above 0 and less than 1, so we added or subtracted 0.001 respectively from the observations with these values. We examined Poisson and negative binomial errors for the count data depending on the dispersion of the data and model fit. Hill Shannon diversity (H') was calculated using the *hill_taxa* function from the hillR package (Daijiang 2018). In the diversity models, gamma and gaussian distributions were

analyzed. Since the gamma distribution does not allow zeros, we added 0.001 to each value before running the models.

In situations where unequal sampling occurred, such as the hours a pan trap was set, an offset was used. An offset standardizes data that were collected with unequal effort (Foster & Bravington, 2013). If an offset was used in a model with a log link, then the offset was log transformed as well. We used multimodal inferences using the corrected AIC (AICc) to select appropriate spatial scales and to choose among alternate models. Prior to modeling, Spearman correlations were performed to identify possible multicollinearity between variables. We used a base model including sampling day-of-year, ambient air temperature, max and average wind speeds, cloud cover, and an offset for pan trap models. We used the *dredge* function from the MuMIn package (Bartoń 2020) and retained significant variables from base models with a $\Delta\text{AIC} < 2$. This model was used as the base mode for future models that included base plus local and base plus landscape variables at each spatial scale. Local variables were measured at each sampling location and included the blooming species richness and percent ground cover of forbs, grasses, and bare ground (described in section 3.3.8). Landscape variables were measured at six spatial scales from 0.5–3 km (0.5, 1, 1.5, 2, 2.5, and 3 km) and included the percent landcover of semi-natural, urban, rewarding agriculture, and nonrewarding agriculture as well as the edge density and landscape Shannon diversity of each spatial scale (described in section 1.8).

Separate models were created for the local variables and the landscape variables at the six spatial scales. Once the best spatial scale was selected, we again used the *dredge* function from the MuMIn package (Bartoń 2020) and selected the

significant variables from models with $\Delta AIC < 2$. Variance inflation factors (VIF) were calculated using the *vif* function from the *car* package (Fox & Weisberg, 2019) and variables were excluded if $VIF > 10$ (Marquardt 1970). P-values for treatment, management, and interaction effects were calculated using the *Anova* function from the *car* package (Fox & Weisberg, 2019). We used Tukey's honest significance difference test within the *glht* function to compare differences among treatment groups (Hothorn et al., 2008). We also used the *emmeans* function from the *emmeans* package to test for differences among management and treatment groups if there was a significant interaction between these variables (Lenth 2022).

To test how the environment may drive functional traits of ground beetle and bee communities, we calculated the fourth corner statistic using the *ade4* package in R (Dray & Dufour, 2007). Local and landscape variables included those described above as well as treatment and management. We selected the landscape scale that each insect group responded to best in the insect abundance GLMMs outlined above (500m for both ground beetles and bees). Traits analysis was not conducted for syrphids due to the abundance of *Toxomerus marginatus* (Say, 1823) ($n=1110$; 91.3% of syrphid captures) and low capture rates of other syrphid species overall. Species traits for the ground beetles included activity (diurnal, nocturnal, or cathemeral), diet (carnivorous, omnivorous, or herbivorous), flight, ground dispersal, and average body size (continuous). Flight and ground dispersal were integer values (1–4), with low values (one) indicating inability of flight/ ground dispersal and high values (four) indicating strong flight/ ground dispersal. Bee functional traits included whether the species is ground-nesting or not (binomial), lecty (integer 1–2), and mean intertegular span (mm)

(Cane, 1987; Gibbs et al., 2023). A lecty value of one indicated that the species is oligolectic and a value of two indicated that the species is polylectic. Since male and female bees often differ in size (Charles Duncan Michener 2007), females are typically better pollinators (Ne'eman et al. 2006), and due to the low proportion of males in our dataset (91 of 600 bees collected from nets), we analyzed bee functional traits of females only. Insect functional traits were extracted from scientific literature (Lindroth, 1961, 1963, 1966, 1968, 1969a, 1969b; Laroche and Larivière, 2003; Holliday et al., 2014; Gibbs et al., 2023). In rare cases when trait information was missing for a species, we estimated the traits based on expert opinion and comparison with related species. We used the FDR correction for multiple comparisons and seeds were set prior to running the analysis for reproducibility. We used the FD package to calculate functional diversity and dispersion of ground beetles (Laliberté & Legendre, 2010). We used the same traits outlined for the fourth corner statistic above to calculate functional diversity and dispersion for each site. We then used GLMMs from the glmmADMB package to test for differences in functional diversity and dispersion among treatment groups (Fournier et al. 2012).

3.4 RESULTS

3.4.1 Vegetation differences among treatments

Forb cover was significantly different among treatment groups (GLMM, $df=2$, $X^2=26.44$, $p<0.001$, Figure 3.2). The percent forb cover was roughly three times greater in the strip (GLMM, 66.7 ± 5.7 %, mean $\pm$ SE) than the control (GLMM, 18.5 ± 2.1 , $df=1$, $z=4.94$, $p<0.001$) and semi-natural treatments (GLMM, 21.2 ± 2.9 , $df=1$, $z=4.05$, $p<0.001$). Likewise, the percent of blooming floral cover was significantly different among treatment groups (GLMM, $df=2$, $X^2=10.22$, $p=0.006$, Figure 3.2). Percent blooming floral cover was over three times greater in the strip (GLMM, 28.3 ± 0.4) than the control (GLMM, 5.8 ± 0.03 , $df=1$, $z=3.19$, $p=0.004$) or the semi-natural treatments (GLMM, 8.3 ± 0.08 , $df=1$, $z=2.65$, $p=0.021$). Control and semi-natural treatments did not differ in the amount of blooming cover (GLMM, $df=1$, $z=1.07$, $p=0.528$). Blooming species richness was significantly different among treatment groups (GLMM, $df=2$, $X^2=16.84$, $p<0.001$). The number of blooming species per quadrat was roughly nine times greater in the strip (GLMM, $1.8 \pm 3.59e-06$) than the control treatments (GLMM, $0.2 \pm 1.07e-06$, $df=1$, $z=3.95$, $p<0.001$). The number of blooming species was not different between the strip and the semi-natural (GLMM, $0.4 \pm 1.83e-06$, $df=1$, $z=1.97$, $p=0.11$) or the control and the semi-natural treatments (GLMM, $df=1$, $z=1.11$, $p=0.49$). The three most commonly identified blooming plants (Table S 7.2) from the strips were sown species, including *Fagopyrum esculentum* ($4.2 \pm 0.7\%$ cover), *Medicago sativa* (L.) ($3.6 \pm 0.79\%$ cover), and *Phacelia tanacetifolia* (Benth) ($1.5 \pm 0.29\%$ cover). However, we did find some volunteer species such as *Cirsium arvense* (L.) ($0.7 \pm 0.27\%$ cover), *Solidago canadensis* (L.) ($0.6 \pm 0.17\%$ cover), and *Sonchus asper* (L.) ($0.5 \pm 0.17\%$ cover). The

percent cover of grasses was significantly different among treatments (GLMM, $df=2$, $X^2=55.67$, $p<0.001$, Figure 3.2). The grass cover within the strip (GLMM, 8.40 ± 1.02) was roughly nine times lower than the control (GLMM, 78.27 ± 1.74 , $df=1$, $z=-7.22$, $p<0.001$) and the semi-natural treatments (GLMM, 72.35 ± 2.70 , $df=1$, $z=-6.14$, $p<0.001$). Control and semi-natural treatments did not differ in their grass cover (GLMM, $df=1$, $z=-0.72$, $p=0.75$). The percent bare ground cover was marginally different among treatments (GLMM, $df=2$, $X^2=6.31$, $p=0.043$, Figure 3.2). The strip had roughly twice as much bare ground (GLMM, $26.16 \pm 1.78e-08$) than the control treatment (GLMM, $12.17 \pm 3.14e-09$, $df=1$, $z=2.41$, $p=0.042$). The amount of bare ground did not differ between the strip and the semi-natural (GLMM, $12.62 \pm 5.02e-09$, $df=1$, $z=2.04$, $p=0.103$) or the control and the semi-natural treatments (GLMM, $df=1$, $z=0.11$, $p=0.993$).

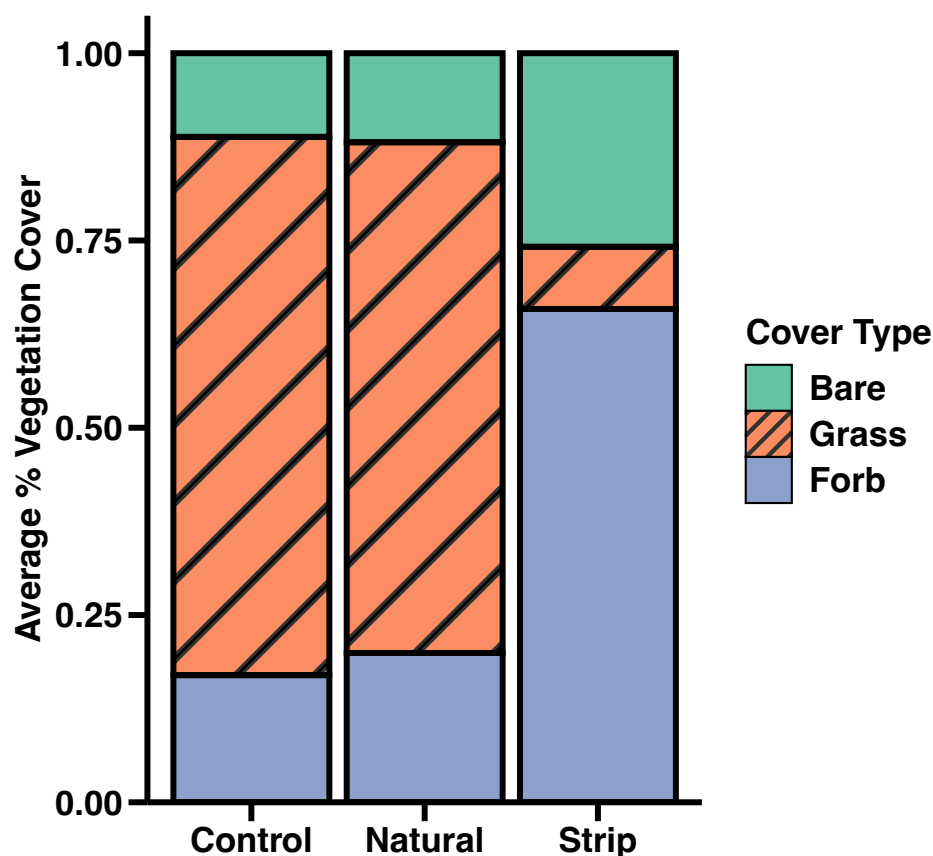


Figure 3.2 Percent cover of forbs, grasses, and bare ground at control, semi-natural, and strip sites sampled in the summer of 2019. Sites included control, locations adjacent to crops without any enhancements; semi-natural, semi-natural areas without management; and strip, locations adjacent to crops with floral strip. Cover types include the percent ground cover of forbs, grasses, and bare ground. Figure was created using ggplot in R (R Core Team, 2023; Wickham, 2016). Sample sizes: strip, $n=15$; control, $n=26$; semi-natural, $n=14$.

3.4.2 Insect data

We collected 4,123 ground beetles in pitfall traps and 600 bees in nets. Pan traps collected 1215 syrphids and 348 bees (Table S 7.3). The most common ground beetle species collected across all sites were *Bembidion quadrimaculatum* (L., 1761) ($n=681$; 16.5% of captures), *Amara littoralis* (Dejean, 1828) ($n=665$; 16.1% of captures), and

Pterostichus melanarius (Say, 1823) ($n=525$; 12.7% of captures). Syrphid captures from pan traps were dominated by *Toxomerus marginatus* ($n=1110$; 91.3% of syrphid captures). The most common bees collected from pan traps included *Apis mellifera* (L., 1758) ($n=68$; 19.5% of bee captures), *Melissodes agilis* (Cresson 1878) ($n=54$; 15.5% of bee captures), and *Andrena thaspii* (Graenicher 1903) ($n=47$; 13.5% of bee captures). The most common bee species from nets were *Apis mellifera* (L., 1758) ($n=266$; 44.3% of captures), *Bombus ternarius* (Say, 1837) ($n=95$; 15.8% of captures), and *Bombus rufocinctus* (Cresson, 1863) ($n=43$; 7.1% of captures).

3.4.3 Community composition

The ground beetle communities were significantly different among treatment types (PERMANOVA, $df=2$, $F=1.93$, $p=0.001$, Figure S 7.1) and pairwise comparisons showed significant differences between the strip and the semi-natural treatment (PERMANOVA, $df=1$, $F=2.63$, $p=0.003$) and between the strip and the control (PERMANOVA, $df=1$, $F=2.15$, $p=0.008$). The syrphid community collected in pan traps was significantly different among treatments (PERMANOVA, $df=2$, $F=1.95$, $p=0.012$, Figure S 7.2), although a pairwise test with FDR adjusted p-values showed no significant differences among the treatment groups. The bee communities collected from pan traps (PERMANOVA, $df=2$, $F=1.61$, $p=0.07$, Figure S 7.3) and nets (PERMANOVA, $df=2$, $F=1.39$, $p=0.127$, Figure S 7.4) were not significantly different among treatments. The bee community collected with nets was significantly different than the bee community collected in pan traps (PERMANOVA, $df=1$, $F=3.31$, $p=0.001$, Figure S 7.5). However, the dispersions between the two collection methods were also significantly different

(ANOVA, $df=1$, $F=7.63$, $p=0.008$). So it is unclear whether the significance of the PERMANOVA is from differences in means or dispersions between the two collection methods (Anderson 2017).

3.4.4 Insect abundance and diversity

Ground beetle capture rates were significantly different among treatments (GLMM, $df=2$, $X^2=9.04$, $p=0.012$, Figure 3.3A). The number of beetles per sample in the strip (GLMM, 115.92 ± 45.89 individuals, mean $\pm$ SE) was roughly three times greater than those collected in the control (GLMM, 32.27 ± 11.29 , $df=1$, $z=2.59$, $p=0.024$) and nine times greater than those collected in semi-natural treatments (GLMM, 11.93 ± 3.82 , $df=1$, $z=2.91$, $p=0.009$). Control and semi-natural treatments did not differ (GLMM, $df=1$, $z=-1.51$, $p=0.276$). Syrphid abundance was significantly different among treatments (GLMM, $df=2$, $X^2=14.83$, $p<0.001$, Figure 3.3B). Syrphids were three times more abundant in the control (GLMM, 23.0 ± 6.54) than in the strip treatments (GLMM, 6.13 ± 2.73 , $df=1$, $z=-3.17$, $p=0.004$). Abundance of syrphids in semi-natural sites (GLMM, 13.92 ± 7.53) did not differ from control (GLMM, $df=1$, $z=-2.07$, $p=0.089$) or strip treatments (GLMM, $df=1$, $z=-0.75$, $p=0.720$). Bees collected with pan traps were significantly different among treatment groups (GLMM, $df=2$, $X^2=11.10$, $p=0.004$, Figure 3.3C) with roughly four times more bees in the strip (GLMM, 10.0 ± 5.59) than control treatments (GLMM, 2.35 ± 0.77 , $df=1$, $z=3.09$, $p=0.005$). The semi-natural treatment was intermediate (GLMM, 3.79 ± 1.53) and did not differ from the strip (GLMM, $df=1$, $z=1.06$, $p=0.529$,) or the control treatments (GLMM, $df=1$, $z=1.52$, $p=0.271$).

Abundances of bees collected with nets were significantly different among treatment groups (GLMM, $df=2$, $X^2=20.82$, $p<0.001$, Figure 3.3D). The number of bees collected with nets was 24 times greater in the strip (GLMM, 33.73 ± 10.53) than the control treatment (GLMM, 1.38 ± 0.57 , $df=1$, $z=4.54$, $p<0.001$) and was eight times higher than the semi-natural treatment (GLMM, 4.14 ± 1.51 , $df=1$, $z=2.35$, $p=0.048$). The abundance of bees collected in the control treatment was not different from those collected at the semi-natural treatment (GLMM, $df=1$, $z=1.62$, $p=0.24$).

The Shannon diversity (H') of ground beetles (GLMM, $df=2$, $X^2=0.68$, $p=0.71$, Figure 3.4A), syrphids (GLMM, $df=2$, $X^2=0.03$, $p=0.98$, Figure 3.4B), or bees collected with pan traps (GLMM, $df=2$, $X^2=3.78$, $p=0.15$, Figure 3.4C) was not significantly different among treatment groups. Net-collected bees were significantly affected by treatment (GLMM, $df=2$, $X^2=7.85$, $p=0.02$, Figure 3.4D). Net-collected bee diversity was highest in the strip treatment (GLMM, 3.11 ± 0.54). The control treatment had roughly five times less diversity (GLMM, 0.65 ± 0.23) than the strip (GLMM, $df=1$, $z=2.36$, $p=0.045$) but was not significantly different from the semi-natural treatment (GLMM, 1.81 ± 0.55 , $df=1$, $z=1.51$, $p=0.27$). Bee diversity at strip and semi-natural treatments did not differ (GLMM, $df=1$, $z=0.57$, $p=0.83$).

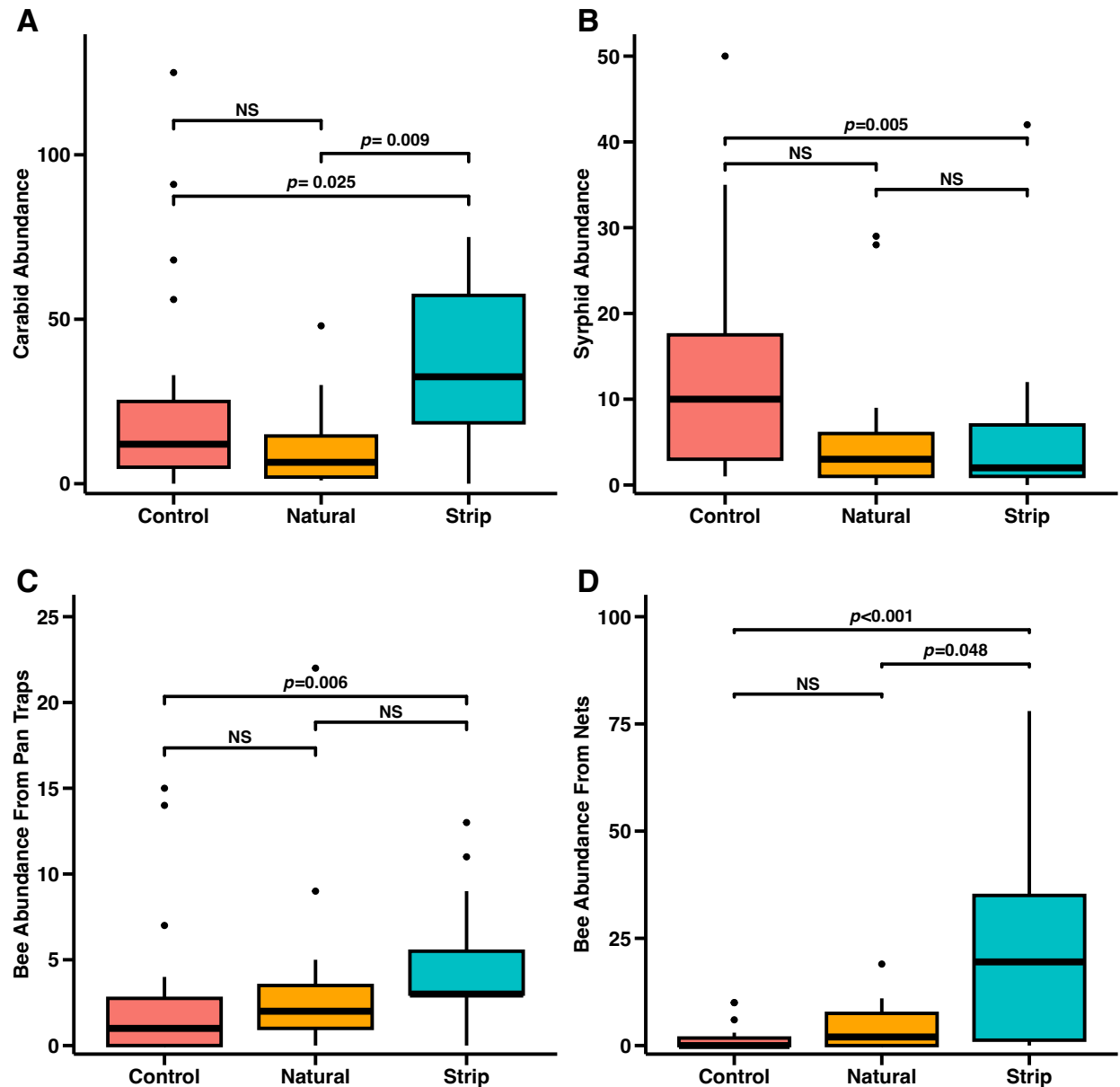


Figure 3.3 Abundance of ground beetles (A), syrphids (B), bees collected with pan traps, (C) and bees collected with nets (D) from control, semi-natural, and strip sites sampled in the summer of 2019 in Manitoba, Canada. Treatments included strip, locations adjacent to crops with floral strips; control, locations adjacent to crops without any enhancements; and semi-natural, semi-natural areas without management. Significance from GLMMs of insect abundance against treatment are shown above the bars. Figure was created using ggplot in R (R Core Team, 2023; Wickham, 2016). Some outliers are not shown on the graph to increase readability. These include (A), Control, 272; Strip, 228, 418, 508 (B), Control, 71, 87, 149; Semi-Natural, 106 (C), Strip, 87 (D), Strip, 139. Sample sizes (A) strip, 12; control, 22; semi-natural, 14. Sample sizes (B, C, D) strip, 15; control, 26; semi-natural, 14.

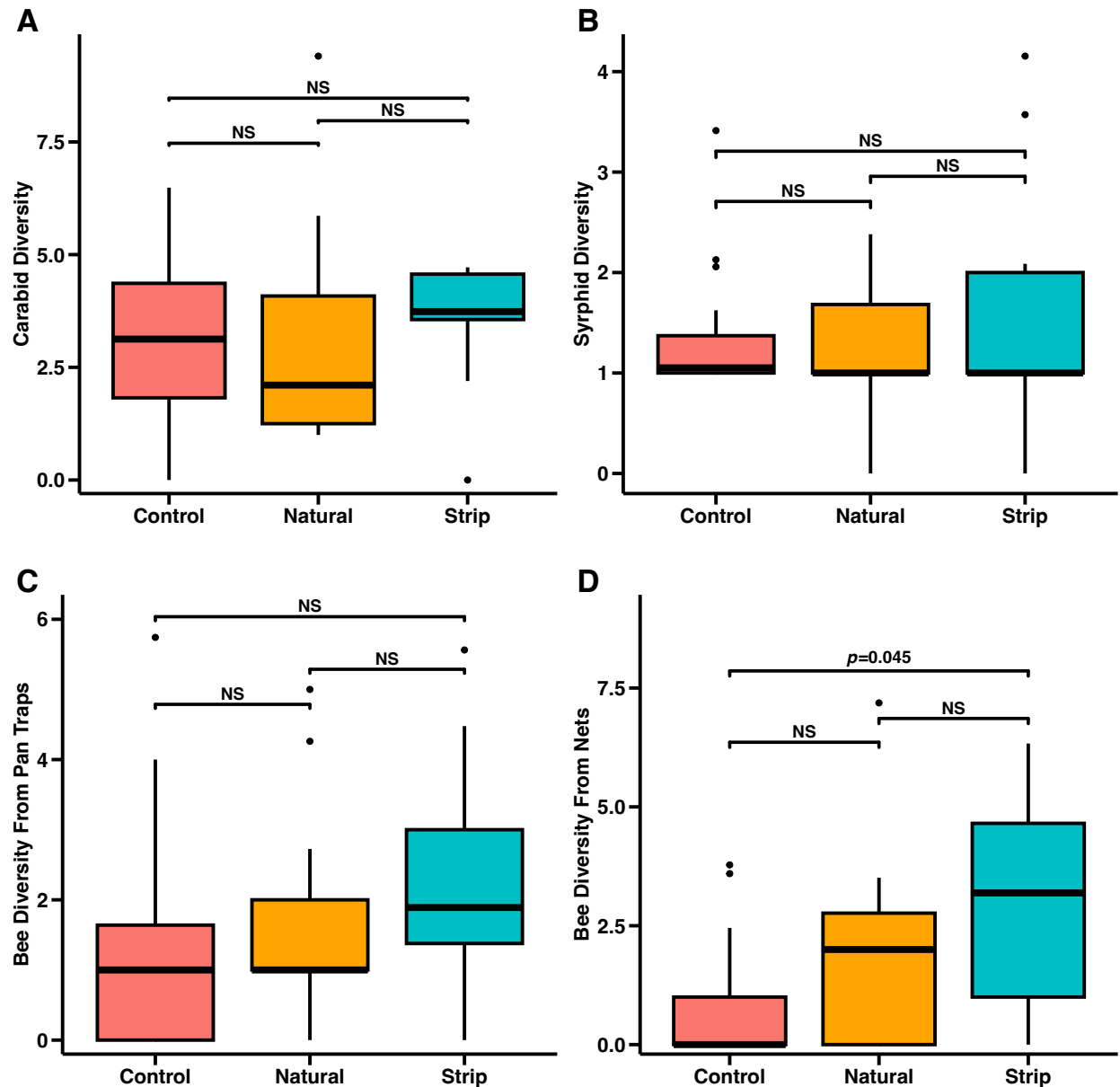


Figure 3.4 Shannon diversity of ground beetles (A), syrphids (B), bees collected with pan traps, (C) and bees collected with nets (D) from control, semi-natural, and strip sites sampled in the summer of 2019 in Manitoba, Canada. Treatments included strip, locations adjacent to crops with floral strips; control, locations adjacent to crops without any enhancements; and semi-natural, semi-natural areas without management. Significance from GLMMs of insect abundance against treatment are shown above the bars. Figure was created using ggplot in R (R Core Team, 2023; Wickham, 2016). Sample sizes (A) strip, $n=12$; control, $n=22$; semi-natural, $n=14$. Sample sizes (B, C, D) strip, $n=15$; control, $n=26$; semi-natural, $n=14$.

3.4.5 Management effects

There was no significant difference between insect communities collected in organic and conventionally managed sites for ground beetles (PERMANOVA, $df=1$, $F=1.25$, $p=0.20$), syrphids (PERMANOVA, $df=1$, $F=2.18$, $p=0.08$), bees collected with pan traps (PERMANOVA, $df=1$, $F=1.23$, $p=0.24$), or bees collected with nets (PERMANOVA, $df=1$, $F=1.13$, $p=0.31$). Similarly, we found no direct effect of management on abundances of ground beetles (GLMM, $df=1$, $X^2=1.36$, $p=0.24$), syrphids (GLMM, $df=1$, $X^2=2.32$, $p=0.13$), bees collected with pan traps (GLMM, $df=1$, $X^2=0.29$, $p=0.59$), or bees collected with nets (GLMM, $df=1$, $X^2=0.12$, $p=0.73$). There was no significant interaction between management and treatment for the abundance of bees collected with pan traps (GLMM, $df=1$, $X^2=1.47$, $p=0.23$) or nets (GLMM, $df=1$, $X^2=0.38$, $p=0.54$). However, we did find that there was a significant interaction between management and treatment effects for ground beetles (GLMM, $df=1$, $X^2=12.43$, $p<0.001$) and syrphids (GLMM, $df=1$, $X^2=6.0$, $p=0.014$). Ground beetles were 1/21 as abundant in conventional control sites (GLMM, 8.23 ± 1.96) than conventional strip sites (GLMM, 177.83 ± 91.42 , $df=5$, $T=5.02$, $p<0.001$), 1/6 as abundant in organic control sites (GLMM, 56.31 ± 20.76 , $df=5$, $T=3.92$, $p=0.002$), and 1/7 as abundant in organic strip sites (GLMM, 62.86 ± 28.88 , $df=5$, $T=3.49$, $p=0.007$). Syrphids were ten times more abundant in conventional control sites (GLMM, 32.69 ± 11.05) than conventional strip sites (GLMM, 3.0 ± 1.18 , $df=5$, $T=4.11$, $p=0.001$) and three times more abundant than organic strip sites (GLMM, 8.88 ± 4.97 , $df=5$, $T=3.1$, $p=0.019$). There were no significant direct effects of management on the H' of ground beetles (GLMM, $df=1$, $X^2=0.93$, $p=0.16$), syrphids (GLMM, $df=1$, $X^2=0.03$, $p=0.86$), bees collected with pan traps

(GLMM, $df=1$, $X^2=0.09$, $p=0.77$), or bees collected with nets (GLMM, $df=1$, $X^2=0.10$, $p=0.75$). We found no significant interaction between management and treatment for the H' of ground beetles (GLMM, $df=1$, $X^2=2.18$, $p=0.14$), syrphids (GLMM, $df=1$, $X^2=0.40$, $p=0.53$), bees collected with pan traps (GLMM, $df=1$, $X^2=1.02$, $p=0.31$), or bees collected with nets (GLMM, $df=1$, $X^2=0.17$, $p=0.68$).

3.4.6 Local and landscape effects

At the local scale, ground beetle captures declined with increasing percent grass cover (GLMM, $df=1$, $X^2=12.25$, $p<0.001$, Table 3.1). The best fitting model for ground beetles was at the 0.5 km scale, where they responded positively to the edge density at 0.5 km (GLMM, $df=1$, $X^2=10.90$, $p<0.001$). Syrphids did not respond significantly to any local variables, but the 1 km spatial scale best explained syrphid catch rates. Syrphids were negatively associated with the percent of semi-natural areas at 1 km scale (GLMM, $df=1$, $X^2=9.33$, $p=0.002$). Bees collected with pan traps responded negatively to the percent bare ground at the local scale (GLMM, $df=1$, $X^2=6.56$, $p=0.01$). The 2.5 km spatial scale best explained capture rates of bees from pan traps, although, bees were not significantly associated with any variables at this scale. Bees collected with nets responded positively to the percent of blooming cover at the local scale (GLMM, $df=1$, $X^2=16.68$, $p<0.001$, Table 1). The 0.5 km scale best explained bee capture rates, but bees were not significantly associated with any landscape variables.

The H' of ground beetles was positively associated with the percent bare ground at the local scale (GLMM, $df=1$, $X^2=5.26$, $p=0.022$, Table 3.2). Ground beetle diversity was best explained at the 3 km scale where they were negatively associated with the

percent of nonrewarding agriculture (GLMM, $df=1$, $X^2=7.85$, $p=0.005$). The H' of syrphids collected with pan traps was not significantly associated with any local variables but was most associated with the 1.5 km spatial scale. Syrphid diversity responded negatively to the percent of semi-natural at 1.5 km (GLMM, $df=1$, $X^2=5.07$, $p=0.024$). The H' of bees collected with pan traps was not significantly associated with any local scale variables but was negatively associated with the amount of urban areas at the 0.5 km scale (GLMM, $df=1$, $X^2=5.0$, $p=0.025$). The H' of bees collected with nets was not significantly associated with any local or landscape scale variables. The best fitting model of the H' of bees collected with nets was at the 2 km scale.

Table 3.1 Local and landscape variables included in the final GLMM models for the abundance of ground beetles, syrphids, and bees collected with both pan traps and nets. Data were collected in Manitoba, Canada in the summer of 2019. Statistics were calculated using the glmmADMB package in R (Fournier et al. 2012; R Core Team 2023). Model terms consist of insect abundance (A), treatment (T), numerical day of year (DOY), offset of pan trap sampling hours (HRS), site level cloud cover (CC), percent local grass cover (GR), percent local bare ground cover (BAR), percent local blooming forb cover (BLM), landscape level percent semi-natural areas (NAT), and landscape level edge density (ED). Associations for significant variables (positive or negative) are given in parenthesis following the base model. All models included site as a random effect. Significance of variables is indicated by the variable name in the Model column: * $p < 0.05$, ** $p < 0.01$, *** $p < 0.001$.

Taxa	Spatial Scale	Model	DF	Log-lik	AIC	VIF
ground beetles	local	A ~ T (-) DOY** (-) GR***	49	-224.824	463.6	1.02 (T); 1.01 (DOY); 1 (GR)
	0.5 km	A ~ T (-) DOY** (+) ED***	49	-225.268	464.5	1.08 (T); 1.01 (DOY); 1.08 (ED)
syrphids	local	A ~ T + HRS (-) CC (sunny)***	52	-186.049	384.1	1.02 (T); 1.02 (CC)
	1 km	A ~ T + HRS (-) CC (sunny)*** (-) NAT*	51	-181.71	377.4	1.01 (T); 1.02 (CC); 1.02 (NAT)
bees (pan trap)	local	A ~ T + HRS (-) BAR*	52	-124.442	260.9	1.01 (T); 1.01 (BAR)
	2.5 km	A ~ T + HRS	53	-131.067	272.1	—
bees (net)	local	A ~ T (+) BLM***	52	-126.883	265.8	1.03 (T); 1.03 (BLM)
	0.5 km	A ~ T + HRS	53	-131.067	272.1	—

Table 3.2 Local and landscape variables included in the final GLMM models for the Shannon diversity (H') of ground beetles, syrphids, and bees collected with both pan traps and nets. Data were collected in Manitoba, Canada in the summer of 2019. Statistics were calculated using the glmmADMB package in R (Fournier et al. 2012; R Core Team 2023). Model terms consist of insect Shannon diversity (H'), treatment (T), numerical day of year (DOY), offset of pan trap sampling hours (HRS), percent local bare ground cover (BAR), landscape level percent semi-natural areas (NAT), landscape level percent non-rewarding agriculture (NRA), and landscape level percent urban areas (URB). Associations for significant variables (positive or negative) are given in parenthesis following the base model. All models included site as a random effect. Significance of variables is indicated by the variable name in the Model column: $*p < 0.05$, $**p < 0.01$, $***p < 0.001$.

Taxa	Spatial Scale	Model	DF	Log-lik	AIC	VIF
ground beetles	local	$H' \sim T (-) DOY^{**} (+) BAR^*$	49	-103.335	220.7	1 (T); 1 (DOY); 1 (BAR)
	3 km	$H' \sim T (-) DOY^{***} (-) NRA^{**}$	49	-102.184	218.4	1 (T); 1 (DOY); 1 (NRA)
syrphids	local	$H' \sim T + HRS$	53	-67.492	145.0	–
	1.5 km	$H' \sim T + HRS (-) NAT^*$	52	-65.225	142.4	1.04 (T); 1.04 (NAT)
bees (pan trap)	local	$H' \sim T + HRS$	53	-58.793	127.6	–
	0.5 km	$H' \sim T + HRS (-) URB^*$	52	-56.719	125.4	1.08 (T); 1.08 (URB)
bees (net)	local	$H' \sim T$	53	6.762	-3.5	–
	2 km	$H' \sim T$	53	6.762	-3.5	–

3.4.7 Capture rates within the field

Ground beetle capture rates did not change as distance from the field edge increased (GLMM, $df=1$, $X^2=0.13$, $p=0.71$) or between strip and control sites (GLMM, $df=1$, $X^2=0.003$, $p=0.96$). Mean capture rates of carabids from pitfall traps within the field was 9.16 ± 0.83 (mean $\pm$ SE) carabids per pitfall trap. There was no significant interaction between distance and treatment on ground beetle abundance (GLMM, $df=1$, $X^2=0.08$, $p=0.78$). Ground beetle H' did not change as distance from the edge of the field increased (GLMM, $df=1$, $X^2=0.03$, $p=0.85$) or between treatments (GLMM, $df=1$, $X^2=0.02$, $p=0.89$). There was no significant interaction between distance and treatment for ground beetle H' (GLMM, $df=1$, $X^2=2.44$, $p=0.12$).

Within field abundance of syrphids collected with pan traps did not change as distance from the field edge increased (GLMM, $df=1$, $X^2=3.58$, $p=0.058$) or between strip and control sites (GLMM, $df=1$, $X^2=0.04$, $p=0.84$). Mean capture rates of syrphids with pan traps within the field was 2.14 ± 0.33 (mean $\pm$ SE) syrphids per pan trap station. There was no evidence of an interaction between treatment and distance from the field edge for abundance of syrphids collected with pan traps (GLMM, $df=1$, $X^2=0.82$, $p=0.37$). Within field H' of syrphids did not change as distance from the field edge increased (GLMM, $df=1$, $X^2=0.49$, $p=0.49$) or between strip and control sites (GLMM, $df=1$, $X^2=0.06$, $p=0.80$). Similarly, there was no evidence of an interaction between distance from the field edge and treatment for the H' of syrphids collected with pan traps (GLMM, $df=1$, $X^2=0.23$, $p=0.63$).

Lastly, within field abundance of bees collected with pan traps did not change as distance from the field edge increased (GLMM, $df=1$, $X^2=1.16$, $p=0.28$) or between treatments (GLMM, $df=1$, $X^2=0.002$, $p=0.97$). Mean capture rates of bees with pan traps within the field was 0.55 ± 0.086 (mean $\pm$ SE) bees per pan trap station. There was no evidence of an interaction between distance and treatment for abundance of bees collected with pan traps (GLMM, $df=1$, $X^2=0.002$, $p=0.97$). Within field H' of bees collected with pan traps did not change as distance from the field edge increased (GLMM, $df=1$, $X^2=2.81$, $p=0.09$) or between strip and control fields (GLMM, $df=1$, $X^2=0.20$, $p=0.66$). There was no evidence of an interaction between treatment and distance from the field edge for the H' of bees collected with pan traps (GLMM, $df=1$, $X^2=0.003$, $p=0.95$).

3.4.8 Functional traits

The fourth corner analysis did not indicate any significant relationships among species traits and environmental traits for the ground beetles or bees. Similarly, no differences were detected among treatment groups for functional diversity (GLMM, $df=2$, $X^2=0.75$, $p=0.69$, Figure 3.5A) or functional dispersion (GLMM, $df=2$, $X^2=0.68$, $p=0.71$, Figure 3.5B) for ground beetles or functional diversity (GLMM, $df=2$, $X^2=0.14$, $p=0.93$, Figure 3.5C) or functional dispersion (GLMM, $df=2$, $X^2=0.18$, $p=0.91$, Figure 3.5D) for bees.

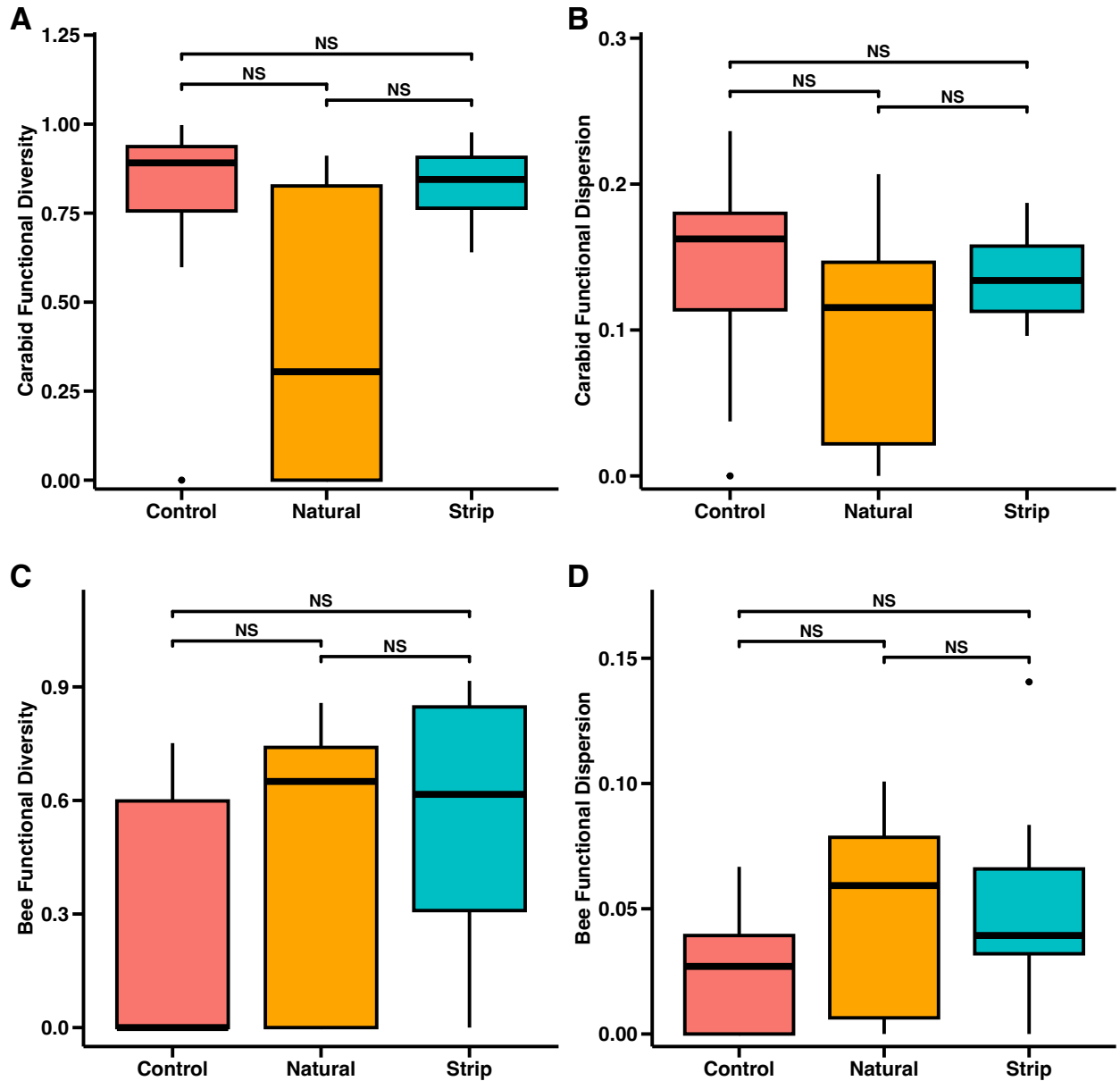


Figure 3.5 Functional diversity (A and C) and functional dispersion (B and D) of ground beetles (A and B) and netted bees (C and D) from control, semi-natural, and strip sites sampled in the summer of 2019 in Manitoba, Canada. Treatments included strip, locations adjacent to crops with floral strips; control, locations adjacent to crops without any enhancements; and semi-natural, semi-natural areas without management. Significance from GLMMs of functional index value against treatment are shown above the bars. Figure was created using ggplot and functional dispersion and diversity indices were calculated using the FD package in R (Laliberté & Legendre, 2010; R Core Team, 2023; Wickham, 2016). Sample sizes (A and B) strip, $n=12$; control, $n=22$; semi-natural, $n=14$. Sample sizes (C and D) strip, $n=12$; control, $n=5$; semi-natural, $n=7$.

3.5 DISCUSSION

3.5.1 Establishment of strips

Floral strips established well overall, despite 2019 being an exceptionally dry and warm summer. We significantly increased both blooming and non-blooming forb cover and decreased grass cover within the floral strips when compared to unmanaged control sites. Other multiyear studies showed increasing abundances of blooming forb species (Blaauw and Isaacs, 2014) and we plan to continue sampling to verify the long-term establishment of our floral strips.

3.5.2 Do floral strips attract beneficial insects?

Capture rates of ground beetles and bees were significantly higher within the floral strips—a finding that is supported by others (Blaauw and Isaacs, 2015, 2014; Campbell et al., 2017; Kordbacheh et al., 2020; MacLeod et al., 2004). Beetle banks in the United Kingdom attract beetles within a matter of months (MacLeod et al., 2004; M. B. Thomas et al., 1991). We found that capture rates of beetles were significantly higher in the floral strips roughly two months after establishment. Pitfall traps measure the activity-density of ground beetles, such that capture rates depend on vegetation structure, local beetle abundance, and activity (Hohbein & Conway, 2018). So, it remains unclear if floral strips had greater beetle abundance, but it is clear that strips increased beetle activity-density when compared to other treatments. Activity density of beetles has been directly linked with ecosystem services (Honek et al., 2003) so we expect that increases in beetle captures rates will coincide with increased ecosystem services in our study as

well. We found significantly more bees in the floral strips when using both pan traps and nets as a sampling technique. These findings are supported by previous studies (Blaauw and Isaacs, 2014; Campbell et al., 2017) that found increased abundances of bees in floral strips. Although the capture rates of bees per sample was low, our capture rates (bees/minute) of netted bees are similar to those observed in the first year of Blaauw and Isaacs (2014).

Syrphid flies were also expected to respond positively to the addition of floral strips (Blaauw and Isaacs, 2014; Haenke et al., 2009), but we found roughly 3.7 times more syrphids in the control fields than the floral strips. It is unclear exactly why syrphids respond this way in our study, but this trend is likely driven by one species, *Toxomerus marginatus*, since this species accounted for over 90% of our syrphid captures. *Toxomerus marginatus* is a widespread species across North America (Skevington et al. 2019) and is commonly collected in other studies (Samaranayake & Costamagna, 2019; Tooker et al., 2006). It is not surprising that we collected this species in such high abundance, but we are unsure why this species was more abundant within the grassy field margins. Some studies have reported that pan trap capture rates of bees may show an inverse relationship with floral abundance (Baum & Wallen, 2011) partially due to the availability of actual foraging options nearby (Cane et al., 2000). It is important to note that we did not identify an inverse relationship between bee capture rates from pan traps and floral abundance, but these observations in bees may explain why syrphid captures were significantly lower in floral strips compared to control fields. Aphidophagous syrphid larvae can consume hundreds of aphids in their lifetime (Curran 1920) and may only disperse 200 metres in agricultural landscapes (Wratten et al. 2003).

Thus, it may be possible that the syrphids are responding to aphids in grassy margins or using them as overwintering habitat (Griffiths et al. 2008). However, the drivers of syrphid abundance in our study are not clear and more research is necessary to better understand these trends.

We found no evidence that strips affected the capture rates of beetles or bees in the adjacent field. Even though capture rates of ground beetles and bees were significantly higher within floral strips than the other treatments, they were not less abundant in the adjacent crop field. This suggests that increases in insect abundances within the strips during early establishment does not result in lower abundances of beneficial insects within the adjacent fields. It may take multiple years for floral strips to fully establish insect communities (Albrecht et al., 2021) so spillover from the strips into the adjacent field may occur over time. We plan to continue to sample these strips to monitor the long-term trends in insect establishment and any possible spillover.

We found that the semi-natural sites held intermediate abundances of bees, but lower amounts of beetles than control sites. We expected capture rates of insects in the semi-natural treatment to be intermediate since undisturbed semi-natural sites should provide better overwintering habitat than control sites (Griffiths et al. 2008). Although bees responded this way, capture rates of beetles were lowest in the semi-natural sites. The three dominant species of ground beetles in our study (*Amara littoralis*, *Bembidion quadrimaculatum*, and *Pterostichus melanarius*) made up 45% of the total captures and have been associated with cultivated lands in Manitoba (Holliday et al., 2014; Luff, 2002). *Amara littoralis* and *B. quadrimaculatum* were both more abundant within the strip treatments ($n= 507$ and 306 respectively) than the control ($n= 32$ and 66

respectively) or the semi-natural treatments ($n= 2$ and 46 respectively), so these species may be driving these trends. *Pterostichus melanarius* was rather abundant within the strip ($n=449$), control ($n=635$), and semi-natural treatments ($n=364$), so it appears that this species may be widespread in the landscape. The three most commonly collected beetles are strongly associated with human activities, commonly found in agricultural areas, and moderately good at dispersing over ground. These species are capable of flight, with the caveat being that *B. quadrimaculatum* is wing dimorphic, so some individuals may be incapable of flight while others show strong flight ability (Holliday et al., 2014; Larochelle & Larivière, 2003). Given that these species favor human activities and have a strong dispersal ability, it is no surprise that they would quickly inhabit a freshly sown floral strip. *Amara littoralis* is a medium-sized, diurnal, omnivorous species, while *B. quadrimaculatum* and *P. melanarius* are mostly nocturnal, carnivorous species. *Bembidion quadrimaculatum* was among the smallest beetles identified in our study (2.8–3.7 mm) while *P. melanarius* was among the largest beetles identified (12–19 mm) (Lindroth, 1969a, 1969b, 1968, 1966, 1963, 1961). These differences in life history traits may result in the community level differences for ground beetles identified in our study, since differences in diet and size likely mean that these species are responding to different rewards within the floral strips.

Floral strips contained a community of ground beetles that was distinct from both the semi-natural and control treatments. Since the most common species of ground beetles were associated with agricultural lands (Holliday et al. 2014), it is most likely that the floral strips were attracting beetles that were already within the fields. The community structure of syrphids and bees collected in this study was not significantly

different among treatments. Other studies have also found that abundances of bees and syrphid flies continued to increase for the first four years after floral establishment (Blaauw and Isaacs, 2014). Thus, it is possible that with continued establishment and sampling of our floral strips, we may see different community level trends for these groups. The most common bee species identified from pan traps in this study were *Apis mellifera*, *Melissodes agilis*, and *Andrena thaspii* while the most common species identified from nets included *A. mellifera*, *Bombus ternarius* and *B. rufocinctus*. It is unsurprising that *A. mellifera* was the most commonly identified bee given its use as a commercial pollinator of canola and other crops (Durán et al. 2010). The most common bees identified from this study are polylectic species, with the exception of *M. agilis*, which is oligolectic on *Helianthus* and related composites (LaBerge 1961). Generalist species are likely responding to the overall increases in blooming forbs from our strip sites while *M. agilis* is likely attracted to *Helianthus annuus* that was included in the seed mix.

Floral strips increase insect diversity over time (Campbell et al., 2017; Kordbacheh et al., 2020) but how quickly this occurs was our particular interest in this study. Although netted bee diversity was significantly higher in the floral strips when compared to the control sites, diversities of all other insect groups were similar among treatments. The close connection of bees to flowers may mean they are early colonizers of floral strip habitats. It is possible that since floral strips were only sampled shortly after establishment that it could take longer for populations of insects to establish in these areas. Ground beetle (MacLeod et al. 2004) and bee (N. M. Williams et al., 2015)

diversities may increase as the floral strips establish. Thus, it is possible that with subsequent years of sampling, we may find differences in insect communities.

None of the local or landscape variables were significantly related to any of the species' traits of ground beetles or bees. Similarly, there were no differences in functional diversity or dispersion among the treatment groups. Thus, it appears that floral strips are effective at attracting beneficial insects shortly after they are established, but the functional composition of the insect communities at strip sites is similar to insect communities of control fields. However, since the strips were only attracting and not retaining insects in this study, it is possible that the floral strips have not been established long enough for insect communities to fully form. We plan to continue to monitor these sites for several years to document any long-term trends in insect establishment and their associated changes in functional insect communities.

In our study, the floral strips were established roughly two months prior to insect sampling. Due to the life cycle of ground beetles and bees, their abundances in the floral strips must be due to the attractive nature of the floral resources and not because the floral strips increased populations of these insects. Since carabids (Thiele 1977) and most wild bees (Fye R. E., 1965; Torchio & Tepedino, 1982) have one generation per year, the insects we collected were likely already born before the floral strips were established. Thus, any effects we see from the floral strips are due to attracting insects rather than supporting or increasing their populations. Even so, other studies with beetles (MacLeod et al. 2004) and bees (Nicholson et al. 2020) have found that the initial increases of beneficial insects sustain over time. From our study, it is clear that floral strips support populations of ground beetles and bees in Manitoba shortly after establishment.

However, studying these floral strips for subsequent years will be able to determine if floral strips can successfully support beneficial insect populations long term.

Due to resource and time limitations, only one sample for each collection method was possible. Additionally, sampling could not begin until the strips had established vegetation, which was not until mid-July (roughly 1.5 months after planting). Due to the large number of sites sampled, and the short collecting season in southern Manitoba, there was not sufficient time to do another round of sampling. Though is recommended by some that pitfall traps be monitored constantly to best sample the ground beetle communities at sites (Luff 2002), our sampling duration of seven days is the most common sampling interval reported in the literature (Hohbein & Conway, 2018). We still managed to collect beetles (O'Rourke et al., 2008) and bees (Blaauw and Isaacs, 2014) in similar abundances to other published studies. However, it is possible that we missed species of insects present at the strips that are active outside of our sampling window, missing to capture the total community present at our sites. Even though we may not represent the entire community of insects here, we still find that this study provides valuable information on the assembly of multiple groups of insects shortly after floral strips are installed.

3.5.3 Local and landscape effects

Each insect group responded differently to local and landscape variables. Abundance of beetles and syrphids were best explained at the 0.5 and 1 km spatial scales respectively, indicating that local resources may be more important than landscape level resources. At the local scale, bee abundance was significantly associated with increased

blooming forb and decreased bare ground cover, indicating that increased forb cover supports higher populations of bees. Bee abundance and diversity was not significantly associated with any landscape variables at scales $>0.5\text{km}$. The diversity of bees collected with pan traps was negatively associated with the amount of urban areas at 0.5 km. This is not surprising as other studies have found significantly less diversity and abundance of bees in urban areas when compared to rural areas (A. J. Bates et al., 2011). Beetle abundance responded positively to the edge density at 0.5 km and beetle diversity responded negatively to the percent of non-rewarding agriculture at 3 km, indicating that increased local heterogeneity and decreased cereal crops in the landscape may support more diverse beetle communities. These results are not surprising given that ground beetles (Purtauf et al. 2005) and bees (Kennedy et al. 2013) have been positively associated with the percent semi-natural areas and a higher diversity of flowering vegetation. Given that ground beetles and bees need overwintering habitats and food resources to reproduce effectively (Cane, 2016; Langellotto & Denno, 2004), the insects collected in our study could be responding to the higher amounts of flowering resources available within the floral strips. However, where beetles and bees responded negatively to nonflowering variables (nonrewarding agriculture, percent grass, and percent bare ground cover) syrphids responded oppositely. Syrphid abundance and richness was negatively associated with the percent of semi-natural areas in the landscape at the 1 and 1.5 km scales, respectively. It is unclear why syrphids responded this way, although, this trend is likely driven by one species, *Toxomerus marginatus*, since this species accounts for over 90% of the syrphid captures in our study.

Management was not a significant predictor of abundance of any insect group and no significant trends were observed between organic and conventional field management. Our results for carabids and syrphids are supported by other studies which found no relationship between insect abundance or diversity and management practices (Moschini et al., 2012; Purtauf et al., 2005). However, we expected to see a trend for bees, since other studies have found more bees on organic sites (Kennedy et al., 2013; Kremen et al., 2002). It is possible that since bees are highly mobile and sometimes have foraging ranges of >1 km (Beil et al., 2008), they are able to find resources (nesting sites, areas with less pesticide exposure, etc.) than less mobile organisms. Thus, local farm management effects may be less important for more mobile insects.

3.6 CONCLUSION

Our study shows that floral strips increase the capture rates of multiple groups of insects shortly after they are established. However, the insect communities of strip sites are functionally similar to control sites and these insects did not spill over into the adjacent strip field. The functional similarities between strip and control sites may be due to the dominance of a few species and these communities may shift as the floral strips establish over time. Although our study only lasted one sampling year, these results indicate there is potential for this practice and we will continue to monitor these sites to determine long term trends. With more research, this practice could help to mitigate the losses of beneficial insects caused by intensive agriculture in the Canadian Prairies.

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CHAPTER 4. FLORAL STRIPS INCREASE THE ABUNDANCE AND SPILLOVER OF BENEFICIAL INSECTS IN ROTATIONAL CROP FIELDS

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4.1 ABSTRACT

1. Insects provide many ecological functions that support agricultural production, but beneficial insect populations are often lower in areas of intensive agriculture. Agronomic practices that support beneficial insect populations are important for sustainable crop production.
2. We installed thirteen floral strips next to agricultural fields in 2019 and then monitored beneficial insects on field edges and within fields 1–3 years after establishment. Treatments included strip, control (crop fields with naturally occurring vegetation), and unmanaged natural sites (no crop or enhancement).
3. Ground beetle active density and Shannon diversity were greater within strips in 2020 but declined to similar levels as control sites in 2021. Though no effect of treatment was found, bees were more abundant in strips compared to other treatments. Both bee abundance and diversity at strip sites increased over time. Syrphid abundance and diversity were not affected by treatment.
4. Spillover of bees into adjacent blooming and non-blooming crops was twice as high in strip fields as the control comparisons. No differences in spillover were found between treatments for syrphids or ground beetles.
5. Floral strips increased the abundance of pollinators within adjacent crops, including blooming canola, which may lead to increased ecosystem services within crops. Our research supports the use of floral strips in annual crop fields to manage local insect populations.

4.2 INTRODUCTION

Regulating ecosystem services, such as pollination and pest control, are valuable for natural and agricultural systems (Morse and Calderone, 2000). Pollinators, including bees (Hymenoptera: Apoidea: Anthophila) and hover flies (Diptera: Syrphidae), directly contribute to reproduction in 90% of plant species (Ollerton, Winfree and Tarrant, 2011). In agricultural systems, pollinators can increase both crop yield (Garibaldi *et al.*, 2013) and crop quality (Klatt *et al.*, 2013), resulting in increased profits for farmers. Hover fly larvae are predators of pest species, like aphids (Curran, 1920), which otherwise can reduce crop yields by up to 40% (Ragsdale *et al.*, 2011). Ground beetles feed on weed seeds, which can reduce weed seed density on fields by up to 55% (Ward *et al.*, 2011), which may ultimately reduce overall herbicide use on farms. Some ground beetles are strictly carnivorous (Forbes, 1883) and feed on insect pests (Grafius and Warner, 1989; Busch, 2016). However, current agricultural practices prioritize high yields over maintaining arthropod mediated ecosystem services (Landis, 2017), which often results in lower populations of insects in agricultural areas (Sánchez-Bayo and Wyckhuys, 2019).

Insects have been in decline for decades. Some long-term studies find trends starting in the 1990s (Hallmann *et al.*, 2017), but pollinator declines from anthropogenic actions dates back at least 50 years (Kevan, 1975). Although the exact reason for these declines varies, it is rarely caused by a single factor (Wagner *et al.*, 2021) and continues even with recent policy changes (Mancini *et al.*, 2023). Agricultural intensification (Hallmann *et al.*, 2017; Outhwaite, McCann and Newbold, 2022), loss of natural areas (Kennedy *et al.*, 2013), insecticides (Kevan, Greco and Belaoussoff, 1997), dietary

changes (Wood *et al.*, 2019), and many other reasons (Potts *et al.*, 2010) often work in tandem to alter insect population size and structure. This is problematic because insect mediated ecosystem services (Klein *et al.*, 2007; Kremen and Miles, 2012) have shown to increase crop quality and yield (Hoehn *et al.*, 2008; Garibaldi *et al.*, 2013) and decrease crop damage from pests (Ahmed, 2023). Given widespread insect declines and the usefulness of beneficial insects for agricultural production, designing agricultural practices for insect conservation should be a priority (Landis, 2017).

Floral strips adjacent to agricultural fields tend to increase beneficial insect abundance (Blaauw and Isaacs, 2014, 2015; Kordbacheh, Liebman and Harris, 2020). They may also increase adjacent crop yield (Blaauw and Isaacs, 2014; Ahmed, 2023) but these responses vary between studies (Albrecht *et al.*, 2021). Increases in insect abundances are not only found on the field edges but may spillover into adjacent fields or habitats (Blitzer *et al.*, 2012). Some studies have found that insect-mediated ecosystem services may reach 200 m into the adjacent fields (Morandin, Long and Kremen, 2014; Woodcock *et al.*, 2016), which may increase yield of adjacent crops if pollination is increased (Holzschuh, Dudenhöffer and Tschardtke, 2012) or pest pressure is decreased (Ahmed, 2023). In Europe, strips used for beetle conservation, called ‘beetle banks’, have been successful at decreasing pest populations within adjacent crops (Collins *et al.*, 2002) and increases in beetle density have been shown to decrease crop damage and herbivory (Ahmed, 2023). Although these strips have proven effective on perennial crops (Blaauw and Isaacs, 2014, 2015; Pywell *et al.*, 2015; MacInnis, Buddle and Forrest, 2020) they may be even more important in areas of annual cropping systems. In areas with crop rotations that may result in years with limited floral resources for bees (cereal crops for

example), floral strips provide an area with permanent food (Graham *et al.*, 2020) and shelter (MacLeod *et al.*, 2004) for beneficial insects—resources which may be lacking in the surrounding landscape. Worldwide over 40% of land is used for agriculture (Outhwaite, McCann and Newbold, 2022). In Manitoba, Canada, almost 50% of available land south of the 51st parallel is used for agricultural crops (Agriculture and Agri-Food Canada, 2020), leaving few natural areas for insects to find their required resources. The area that is used for agriculture is rather continuous with large field sizes (approximately 60 ha). In this region, canola (*Brassica napus* L.) is one of the most commonly planted crops (Agriculture and Agri-Food Canada, 2020) and contributes to over \$13 billion in sales annually across Canada (Statistics Canada, 2023). Although canola is capable of self-pollination, pollination by both managed and wild bees can increase seed production (Morandin and Winston, 2005; Woodcock *et al.*, 2019).

The landscape surrounding floral strips influences their effects on beneficial insects (Tschamntke *et al.*, 2005), so they might not be equally as effective in every landscape that they are placed. High quality habitat (Purtauf *et al.*, 2005; Kennedy *et al.*, 2013) and landscape complexity (Priyadarshana *et al.*, 2024) are positively associated with beneficial insect abundance and diversity. Floral strips may be more important in areas with limited existing floral resources as pollinators responded positively to increases in forb richness (Scheper *et al.*, 2015)- especially in early years of establishment. Although modern agricultural practices maximize field size to increase overall yield, a study from Alberta found that leaving 30% of land out of production within 0.75 km of fields can maximize profits of canola crops (Morandin and Winston, 2006) through increased pollination services (Morandin and Winston, 2005). This is

problematic because natural areas are few and far between in the agricultural production area of Manitoba (Agriculture and Agri-Food Canada, 2019). Given that southern Manitoba is dominated by agricultural landcover types (Agriculture and Agri-Food Canada, 2020) and that insect responses are often inconsistent (Karp *et al.*, 2018) assessing which landscapes are best suited for floral strips here is important to effectively use this management tool. If floral strips are not found to be effective, then supporting the conservation of existing natural areas may be an important management tool.

A previous study of ours during the summer of 2019 found that floral strips increased abundances of ground beetles and bees after initial establishment (Killewald *et al.*, 2024). We only found increases of beneficial insects within the strips themselves as we did not identify any spillover effects into the adjacent crops. We found increases in insect abundance regardless of the surrounding landscape or farm management style. However, we only included one years of data in the previous manuscript due to a change in the study design. In 2019, farmers planted the same crops in the strip and control fields- allowing for comparisons of insect communities and abundances between fields. In 2020 onwards, farmers planted different crops between field types, so we adjusted our sampling protocol accordingly (described in more detail in the methods). The results from our 2020 and later samples are reported and discussed here. We continued to build off our previous work and ask the following questions:

1) Do floral strips continue to increase the abundance and diversity of beneficial insects in years 2–4 of establishment? If so, do these abundances increase over time? 2) Which local (blooming forb richness, diversity, and overall vegetation structure) and

landscape (composition and configuration) level variables effect beneficial insect assemblages (abundance and diversity)? 3) Do beneficial insects spill over into adjacent fields throughout the year and if so, do they continue to spill over during canola anthesis (bloom)? 4) How do the communities of beneficial insects (ground beetles, bees, and syrphids) compare among floral strips, control fields, and semi-natural treatments?

4.3 METHODS

4.3.1 Site selection

Fifteen floral strips were established on the edges of rotationally managed crop fields located in Manitoba, Canada in the summer of 2019 (Figure 4.1). The strips were established on the edges of fields, meaning that the unmanaged grassy border adjacent to the strip was left intact and a small area of the field was taken out of production. Floral strip size was 0.64 ± 0.097 ha (mean $\pm$ SE) in size (range 1.58–0.15 ha) and was 8.0 ± 0.75 m wide (range 3.65–13.4 m). The seed mix for the floral strips was broadcast at a rate of 22.4 kg/ha and lightly harrowed into the soil between 30 May and 6 June 2019. Strips contained a mixture of fifteen species of forbs and four species of grasses (Table S 7.1). Plant species were selected based on availability, cost, establishment success, and attractiveness to insects. Strips were generally > 10 k apart from each other (Killewald *et al.*, 2024). Due to issues with establishment at two sites, only thirteen of the strip sites were sampled from 2020–2022 (Figure 1). Within a 10 km radius of each strip site, we also sampled a paired control field and semi-natural site. A strip, control, and natural site

within a 10 km radius is referred to as a site cluster (or block for statistical analysis). All locations within a site cluster were sampled on the same day except in very rare ($n=2$) circumstances where sampling was completed the following day. Control sites were rotationally managed crop fields that did not contain any enhancements or floral strips. Sampling at control fields took place on unmanaged, grassy borders of the field. Semi-natural (natural) sites were locations that were unmanaged for at least five years prior to sampling. Often, these were forest edges or riparian areas but, in some cases, ($n=1$) natural sites were edges of old pastures. Sites within a cluster were as close as possible, but natural sites were often hard to locate in the landscape. This ultimately pushed our distance between sites in a cluster to a wider radius. Thus, these were often not protected or known to contain high biodiversity since we often selected one of the few (or only) natural areas within the surrounding landscape. Each strip site also contained an on-site control treatment (OSC), which was located at least 150 m away from the floral strip and was placed on an unmanaged, grassy margin of the strip field. This margin contained mostly grasses with volunteer vegetation and acted as a buffer between the crops and other landcover types. The OSC and control margins were generally ~5 metres wide. The OSC and the control treatments were functionally similar—the only difference being that the OSC was located at the same field as the floral strip to account for differences in crop management between fields. Locations sampled in this manuscript were the same locations that were sampled in our previous study (chapter 3), but differences in collection methods between studies are described in each section below. Crops at strip and control fields were planted and managed by the farmers and varied annually. Crops

included barley, canola, corn, flax, oats, potatoes, soybeans, wheat, and intercropped oats/peas.

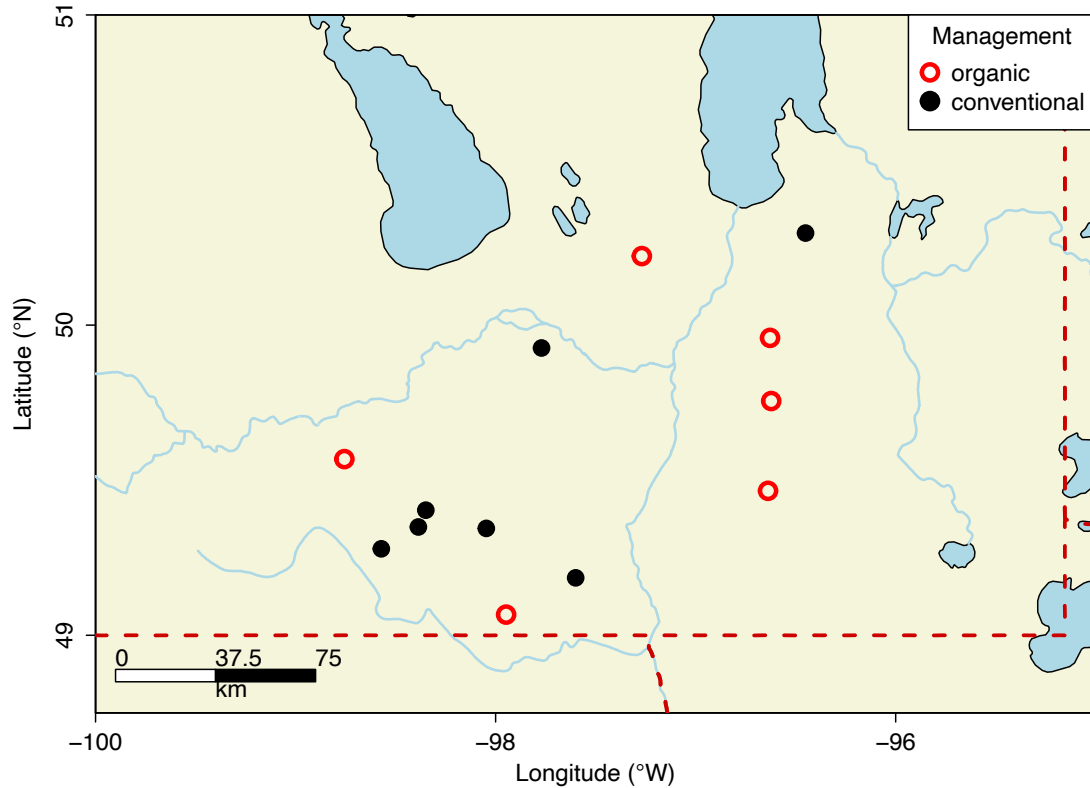


Figure 4.1 Locations of all blocks of sites where beneficial insects were sampled at farms in southern Manitoba during the summers of 2020–2022. All sites were sampled multiple times for each collection method. Dashed red lines represent borders between Manitoba, Ontario, North Dakota and Minnesota. This figure was created using the PBSmapping package in R (R Core Team, 2023; Schnute et al., 2021). Sample sizes: conventional (allows for the use of insecticides, herbicides, and fungicides), $n = 7$; organic (certified organic by the Canadian Organic Standards), $n = 6$.

4.3.2 Pitfall traps

Pitfall traps were deployed in both the field edges and within fields at multiple distances to collect ground beetles. Five pitfall traps were placed within each of the four treatments of edges (strip, OSC, control, and natural). Pitfalls were placed within the middle of each treatment, roughly equidistant between each of the treatment boundaries.

Additionally at the strip and OSC, three pitfall traps were placed at both 15 and 50 m away from the edges of each treatment. In total, 32 traps were deployed per site cluster—11 pitfall traps at the strip and OSC and five at the control and natural treatments. Pitfall traps were placed five metres apart from each other (Hohbein and Conway, 2018). We used five pitfall traps on the field edges as this matched our previous study and increased the replication of traps within the adjacent field (1 trap was used per distance within the field previously). Additionally, we processed each trap separately here where as we pooled samples based on location in our previous study (chapter 3),.

Each pitfall trap consisted of two 1 L deli cups that were nested within each other with a plastic mesh (2.5 x 2.5 cm square holes) lid to exclude mammals. Each trap was then filled with 250 ml of a 50% propylene glycol and 50% water mixture. Pitfall traps were deployed in times of clear weather (no rain and generally >20 °C during the days) and were left in the fields for 7 days. Samples were collected from the fields by straining insects from the propylene glycol solution and placing them into bags with 70% ethanol. Samples were then brought to the lab where ground beetles were processed (see specimen processing section below). Pitfall traps were deployed in the fields between 7 June and 2 September in both 2020 and 2021. We sampled each site twice per year (mean 2.0 ± 0.17 SE) and processed a total of 1,400 individual pitfall trap samples.

4.3.3 Blue vane traps

Blue vane traps were deployed on field edges at the same time as the pitfall traps to collect bees and syrphids. Blue vane traps used here contained a yellow base and two blue vanes on top (BanfieldBio, Seattle, Washington). Three blue vane traps were placed

within each treatment, generally near the pitfall traps. Thus, blue vane traps were spaced five metres apart and were located roughly in the middle of each treatment. Each trap was then filled with roughly 750 ml of 50% propylene glycol and 50% water solution. Blue vane traps followed the sampling methods described above for sampling length and dates. We sampled each site twice per year (2.0 ± 0.17) and processed a total of 471 individual blue vane trap samples. Blue vane traps were not used in our previous study and were implemented here to increase the capture rates of bees and syrphids.

4.3.4 Pan traps

Pan traps were deployed at the same locations as pitfall and blue vane traps (but at different times) to collect bees and syrphids. Five pan trap stations were deployed in treatment edges and were located five metres apart from each other. Additionally, three pan trap stations were deployed within the fields at 15 and 50 m in front of the strip and OSC treatments ($n=32$ per cluster). Pan traps were placed within the fields throughout the summer, but samples that were collected during canola bloom were analyzed separately. Each pan trap station contained three different colours of pans (one 82 ml pan each of white, blue, and yellow) and were adjusted to the height of the surrounding vegetation on a metal pole. Pan traps were filled with a soapy water solution and were installed in the morning and removed from fields at the end of the sampling day. Deployment time varied (4.42 ± 0.08 hours, mean $\pm$ SE) by site and was used as an offset in generalized linear models to standardize collection effort. Samples were collected by removing insects within each bowl and placing them into a bag with 70% ethanol solution. Each bowl in the pan trap station was processed separately whereas samples were pooled

together based on location in our previous study. Samples were then brought back to the lab for further processing (see specimen processing section below). We sampled each site two or three times per year (2.57 ± 0.2) and processed a total of 5,575 individual pan trap samples. Pan traps were deployed in the fields from 7 June until 18 August in both 2020 and 2021.

4.3.5 Targeted pollinator netting

Bees and syrphids were netted along two transects in each of the four treatment edges. Transects were at least five metres apart from each other and generally started from the midpoint of the treatment and extended parallel to the longest edge. When farmers planted canola adjacent to a strip site, bees were also netted along two transects at both 15 and 50 m away from both the strip and OSC treatments during canola bloom ($n=16$ per cluster during canola bloom, $n=8$ at all other sites). These additional transects were used to determine possible spillover effects of bees during canola bloom. Net transects followed those described in our previous study except the time, length, and abundance (previously $n=1$ transect, 15 minutes, 30 metres) of the transects were changed for this study in order to standardize collection methods within our department (Hanuschuk, 2021; Martini, 2022). Each transect was 50 m long and pollinators were collected for 10 minutes per transect. Timers were paused when an insect was being removed from the net. Insects were separated based on host plant and transect location/number before being brought back to the lab for further processing (see specimen processing section below). Net transects were conducted between 8 June and 4 August each year from 2020 until 2022. We sampled each site one or two times per year

(1.78 ± 0.12) and processed a total of 586 transect samples. Net transects either coincided on the same day as pan trapping, or rarely, on the previous or following day.

4.3.6 Specimen processing

Bees and syrphids were pinned and identified to species while ground beetles were identified to genus due to their abundance and time constraints. Identifications were based on taxonomic literature and were referenced against expertly identified specimens and verified by taxonomists where possible. Ground beetles were identified with keys from Lindroth and Bousquet (Lindroth, 1961, 1963, 1966, 1968, 1969b, 1969a; Bousquet, 2010). Syrphids were identified using Skevington *et al.* (Skevington *et al.*, 2019). Bees were first identified to genus (Packer, Genaro and Sheffield, 2007) and then to species using keys for each respective family: Andrenidae (LaBerge, 1967, 1969, 1971, 1973, 1977, 1980, 1986, 1987, 1989; Ribble, 1967, 1968, 1974; Shinn, 1967; LaBerge and Bouseman, 1970; LaBerge and Ribble, 1972, 1975; Bouseman and LaBerge, 1979; LaBerge and Thorp, 2005), Apidae (LaBerge, 1956b, 1956a, 1961; Lavery and Harder, 1988; Droege *et al.*, 2010; Rehan and Sheffield, 2011; Williams *et al.*, 2014; Droege and Rehan, 2016; Onuferko, 2017, 2018), Colletidae (Stephen, 1954; Snelling, 1966, 1968, 1970, 1983), Halictidae (Sandhouse, 1941; McGinley, 1986, 2003; Coelho, 2004; Gibbs, 2010, 2011; Gibbs *et al.*, 2013), and Megachilidae (Sandhouse, 1939; Michener, 1947; Baker, 1975; Sheffield *et al.*, 2011; Gonzalez and Griswold, 2013; Semmler *et al.*, 2017). Specimen processing largely followed what was described in our previous study (chapter 3), except that ground beetles were pinned and identified to genus.

4.3.7 Floral surveys and weather data

Vegetation surveys and weather data were collected on the same days as the net transects. Ground cover was measured using a 1 x 0.5 m quadrat placed every five metres along each of the net transects described in the targeted pollinator netting section ($n=11$ quadrats per transect). Flowers were identified to genus or species (Scoggan, 1957; Newcomb, 1989) which was then used to describe floral diversity within each transect. The percent cover of each blooming forb species, and total cover of grasses, forbs (blooming and non-blooming), and bare ground was visually estimated for each quadrat using the figures provided by Folk (1951). At each sample, cloud cover (sunny, partly cloudy, or overcast) was determined visually. Average and maximum wind speed and ambient air temperature were recorded with a Kestrel 3000 pocket weather meter (Philadelphia, PA). Weather data were collected because local weather conditions affect insect foraging and activity (Goodwin *et al.*, 2021). These variables were used in base models for later analysis. Floral surveys and weather collection largely followed the methods described in our previous study (chapter 3), but differs based on the length of the vegetation transect.

4.3.8 Landscape analysis

Landscape composition and configuration data of each site and sampling year (2020–2022) combination were extracted from the annual crop inventory data layer from Agriculture and Agri-Food Canada (Agriculture and Agri-Food Canada, 2021) using the *landscapemetrics*, *raster*, and *sampSurf* packages in R (Hesselbarth *et al.*, 2019; Gove, Jeffrey, 2021; Hijmans, 2021; R Core Team, 2023). Data were extracted at six spatial scales (0.5, 1, 1.5, 2, 2.5, and 3 km away from sampling locations) because different

insect groups will likely respond to different spatial scales (Karp *et al.*, 2018). Overall, methods for extracting landscape composition and configuration data large follow those used previously.

Landscape composition data were separated into four categories, semi-natural, developed, and two forms of agriculture (cereals and non-cereals). We separated agricultural crops into the two categories since insects will likely respond differently to these crop types (Montero-Castaño, Ortiz-Sánchez and Vilà, 2016). Non-cereals included crops such as canola, flax, soybeans, peans, beans, and other vegetables. Cereals included crops such as wheat, oats, and barley. Semi-natural landcover included land use types such as forests, shrubland, and wetlands. Lastly, developed lands included those with permanent anthropogenic structures such as buildings, roads, and cities. We extracted the percent cover of each of these categories using the *lsm_c_ca* function from the landscapemetrics package in R (Hesselbarth *et al.*, 2019; R Core Team, 2023). Landscape configuration data were extracted as the edge density and landscape Shannon diversity using the *lsm_l_ed* and the *lsm_l_shdi* functions, respectively, from the landscapemetrics package in R (Hesselbarth *et al.*, 2019; R Core Team, 2023).

4.3.9 Data analysis

4.3.9.1 Data analysis introduction

We constructed multiple generalized linear mixed effect models (GLMM) from the glmmTMB package in R (Brooks, Kristensen and Koen, 2017; R Core Team, 2023) to examine differences in dependent variables (floral composition, insect abundance, and

insect Shannon diversity) between treatment groups, year, distance into the field, and the interaction among these variables (objectives 1 and 3). We also tested how fixed effects (weather, local, and landscape variables) influence beneficial insect responses (objective 2). We compared the strip, control, and natural treatments for models with full replication at the site cluster (that is, they collected comparable data from strip, control, and natural treatments: such as objectives 1, 2, and 4). For models with only partial sampling at the block level, such as objective 3 and objective 4 (which compared data collected within the strip field only or that was only located at strip sites) we included the OSC samples as the control as no samples for these datasets were collected at the separate control fields due to differences in crop management. Models included an offset (Foster and Bravington, 2013) to standardize pan trap collection effort (active collection hours) and a standardized value (1) was used for samples where collection effort was similar (net and blue vane samples). Offset values were included in the dataset as a column and were called into GLMMs directly using the “offset” argument. If an offset was included in a model with a log link, then the offset was log transformed as well. Site cluster (block) was used as a random effect for all models.

We used GLMMs to determine differences in vegetation structure between treatments (strip, control, or natural) as these may help to explain insect responses to strips. Model dependent variables included the percent cover of forbs, grasses, or bare ground, blooming vegetation richness or Shannon diversity. Fixed effects included treatment, year, and the interaction between these two variables. We created three separate models to determine insect responses to floral strips (objective 1) as well as their surrounding environment at local and landscape levels (objective 2).

4.3.9.2 Objective 1: Do floral strips increase the abundance and diversity of beneficial insects over time?

To test if floral strips increase the abundance and Shannon diversity of beneficial insects and how these trends change over time (objective 1), we used a dataset that included all sampling methods for each insect group (pitfall traps for ground beetles and nets, pan traps, and blue vane traps for bees and syrphids) at three treatments (strip, control, and natural). Fixed effects included treatment, year, the interaction between these two variables, and sampling day of year. We also included control variables where possible (for bee and syrphid datasets) such as the ambient temperature and windspeed to account for differences in weather that might influence insect capture rates. We also included an offset to standardize collection effort.

4.3.9.3 Objective 2: Which local and landscape level variables explain beneficial insect responses to floral strips?

To determine which local and landscape variables influence beneficial insect responses (objective 2) we used the same insect dataset and model described above for objective 1. We tested how the dependent variable (insect abundance or Shannon diversity) changes with treatment, year, and the interaction between these variables and included fixed effects described below. We used the same model created in objective 1 as our base model. We then used this in base plus local and base plus landscape models. Because insects will respond at different spatial scales (Karp *et al.*, 2018) we separated

our models into local (which measured variables such as vegetation structure) and landscape (0.5 – 3 km) level variables.

Local models included the final base model described above plus vegetation survey data as fixed effects (percent cover of grasses, bare ground, and both blooming and non-blooming forbs, blooming forb richness, and blooming forb Shannon diversity). We then simplified the models by retaining only significant fixed effects. We used the average annual vegetation data for each site because vegetation data were not collected during the pitfall and blue vane samples (only the net and pan trap samples).

Landscape models included the final base model described above plus both the landscape composition and landscape configuration variables described in the landscape analysis section as fixed effects. We first tested for the best fitting spatial scale by including all landscape variables of a respective spatial scale (0.5, 1, 1.5, 2, 2.5, or 3 km) into its own model and selecting the spatial scale with the strongest response (lowest AIC score and/or highest log likelihood). We then simplified the models by retaining only significant fixed effects.

4.3.9.4 Objective 3: Do beneficial insects spill over into adjacent fields throughout the year and/or during canola anthesis?

We used a dataset that included insect abundance and Shannon diversity on field edges (0 m) and up to 50 m into the fields in front of strip and OSC treatments to determine if beneficial insects spill over into adjacent crops. The OSC was used here instead of the control treatment because farmers planted different crops in the strip and control fields. We created two sets of data– one that contained only data that was

collected on blooming canola fields and one containing data of all crops. Data used here was from the pitfall traps, pan trap, and net transects collected within the fields and on strip and OSC edges. We tested for differences in dependent variables (insect abundance and Shannon diversity) between fixed effects such as distance into the field, treatment, and the interaction between treatment and distance for both data sets.

For all models described above (objectives 1–3), we first checked that the best error distribution was applied by comparing Akaike information criterion (AIC) and log-likelihood scores. For percentage data, such as the vegetation cover dataset, we used a beta error-distribution. We compared Poisson and negative binomial distributions for insect count data and selected the distribution with the best model fit. Unless otherwise stated in the results for count data, the negative binomial distribution was used. Hill Shannon diversity was calculated using the *hill_taxa* function from the *hillR* package in R (Daijiang, 2018; R Core Team, 2023) and data were compared using the Gaussian error distribution. Prior to modeling, Spearman correlations were calculated to identify possible collinearity between pairs of variables. If strong correlations were found ($> \pm 0.7$) then only one variable was retained based on comparative AIC and log likelihood scores. We calculated variance inflation factors (VIF) using the *check_collinearity* function from the *performance* package (Lüdecke *et al.*, 2024) and fixed effects were excluded if $VIF > 10$ (Marquardt, 1970). Chi-square (X^2) and p -values for variables and the interaction between variables were calculated using the *Anova* function from the *car* package (Fox and Weisberg, 2019). We selected *type = "III"* if models included an interaction. We used the *emmeans* and *emtrends* functions from the *emmeans* package

(Lenth, 2022) to test for differences between treatment groups. Effect sizes were calculated using the *standardize_parameters* function with method="posthoc" from the effectsize package (Ben-Shachar, Lüdtke and Makowski, 2020).

4.3.9.5 Objective 4: How do the communities of beneficial insects compare among treatments?

We used ordinations to examine compositional differences in insect communities between treatments (strip, control, and natural) using the data collected for objective 1. We also tested if spillover insect communities were different by comparing edge and within field samples at strip and OSC treatments in both all crops and canola during its anthesis (bloom) only. We used the data collected for objective 3 to determine differences in spillover communities.

Ordinations for each dataset were created using a distance-based redundancy analysis using the *capscale* function with a Bray-Curtis dissimilarity measure and were visualized using the *ordiplot* function from the *vegan* package (Oksanen *et al.*, 2019). We then used a permutational multivariate analysis of variance (PERMANOVA) using the *adonis2* function from the *vegan* package (Oksanen *et al.*, 2019) to test for significant differences between treatment groups or sampling location (field or edge). Comparisons were restricted by clusters (blocks) of sites using the *permutations* argument in the *adonis2* function. If a significant result was detected from the PERMANOVA, then we ran a pairwise comparison of groups with an FDR adjusted *p*-value using the *pairwiseadonis* function from the *pairwiseAdonis* package (Martinez Arbizu, 2017). PERMANOVAS may be sensitive to both differences in group centroids and group

dispersions (or both) (Anderson, 2017) so we tested for differences in group dispersions separately using the *betadisper* and *permutest* functions in the *vegan* package (Oksanen *et al.*, 2019). If this returned a significant output, then we tested for differences in dispersion between individual groups with the *TukeyHSD* function in R (R Core Team, 2023). Differences in group dispersions indicate that individual samples within each respective group have different levels of spread within the multivariate cloud (Anderson, 2017). In other words, if dispersions are different, then community variation within groups is significantly different. While differences in group centroids indicate that the average community composition of groups is different. Unless explicitly stated in the results, group dispersions were not significantly different from each other. We used 999 permutations for all permutational analysis and seeds were set prior to allow for reproducibility.

4.4 RESULTS

4.4.1 Floral strip establishment

Overall floral strips established well as blooming forb richness was three to four times higher in strip sites (Table S 7.4) when compared to control (GLMM, $df=1$, $t=10.61$, $p< 0.0001$) and natural sites (GLMM, $df=1$, $t=9.66$, $p< 0.0001$), regardless of year. Although forb richness differed between treatments, Shannon diversity was similar (Table 4.1).

We found a significant interaction between treatment and year for both the percent forb and grass cover identified from floral surveys (Table 4.1). Strip sites contained the highest percent cover of forbs and bare ground and the lowest grass cover of any treatment across all sampling years (Table S 7.2). Over time, the amount of forb cover at strip sites decreased ($-11.84\% \pm 2.8$ per year) while the percent of grass cover increased ($+14.1\% \pm 2.9$ per year). Forb and grass cover at control and natural sites did not change over time.

Table 4.1 Statistical results of generalized linear mixed effect models of vegetation data collected at sites of three treatments in Manitoba, Canada in the summers of 2020–2022. Treatments included strip, locations adjacent to crops with floral strips; control, locations adjacent to crops without any enhancements; and semi-natural, semi-natural areas without management. Blooming forb H' represents the Shannon diversity of blooming forbs. Rows for forb, grass, and bare ground indicate the percent of ground cover of each category. Site was used as a random effect for all models.

measure	df	<u>treatment</u>		df	<u>year</u>		<u>treatment * year</u>		
		X ²	p		X ²	p	df	X ²	p
blooming forb richness	2, 341	19.17	< 0.0001	1, 342	< 0.01	0.26	2, 341	2.18	0.34
blooming forb H'	2, 341	4.52	0.10	1, 342	< 0.01	0.97	2, 341	1.23	0.54
forb	2, 341	48.4	< 0.0001	1, 342	0.01	0.91	2, 341	12.61	0.002
grass	2, 341	80.52	< 0.0001	1, 342	0.01	0.93	2, 341	17.2	0.0001
bare ground	2, 341	11.16	0.004	1, 342	0.10	0.74	2, 341	1.37	0.50

4.4.2 Insect abundance and composition

We found a total of 25,343 ground beetles from 23 different genera with pitfall traps (Table S 7.5). Of those, a total of 15,441 were found within the fields. The most common ground beetle genera were *Pterostichus* ($n=18,615$), followed by *Harpalus* ($n=2,576$) and *Amara* ($n=1,764$).

Across all sampling methods ($n=3$), we found 12,786 bees from six families (Table S 7.5). Of those, 634 bees were collected from within the fields. We collected 140 species from 28 genera, including the first record of *Eucera lepida* (Cresson) in Manitoba (determined by JG). A single female was collected on 2021-06-21 near Binney Corner Nature Preserve (49.26439, -98.59267) with a blue vane trap. Apidae totaled 80 % of bee captures in our study followed by Halictidae (10.9 %) and Megachilidae (4.4 %). Overall, the most common bee species was the managed *Apis mellifera* L. ($n=4170$, 32.6% of captures). The most common wild bee species included *Bombus borealis* Kirby ($n=2030$, 15.8 %), *Melissodes agilis* Cresson ($n=1084$, 8.4 %) and *Melissodes trinodis* Robertson ($n=1008$, 7.8 %). *Apis mellifera* (43.5 % of strip captures) made up most of the bee captures from strip sites followed by *Bombus borealis* (18.8 %), and *Melissodes agilis* (8.9 %). The most common species from other treatments included the same three species, but *B. borealis* (21.4 %) was more common than *A. mellifera* (20.5 %). *Melissodes agilis* (12.5 %) was still the third most common species from control and natural treatments.

We also collected 3,267 syrphids across all sampling methods (Table S 7.5), of which, 560 were collected within the fields. We collected 38 syrphid species from 16 genera. The most common syrphids were *Toxomerus marginatus* (Say) ($n=1793$),

Toxomerus geminatus (Say) ($n=309$), and *Eupeodes latifasciatus* (Macquart) ($n=302$). We found a total of 9,510 bees and 1,524 syrphids with blue vane traps, 1,813 bees and 729 syrphids with nets, and 1,296 bees and 935 syrphids with pan traps.

4.4.3 Beneficial insect abundance and diversity on field edges

Beneficial insect abundance and Shannon diversity responses differed depending on the taxon (Table 4.2). Carabid abundance and diversity increased in floral strips, but syrphid flies were unaffected. Though no effect of treatment was found for bees, they were twice as abundant within strips as other treatments. The abundance and diversity (Figure 4.2A, B) of carabids collected with pitfall traps on field edges was dependent on both treatment and year as there was a significant interaction between these two variables (Table 4.2). In 2020, carabid abundance (30.7 ± 4.0) and diversity (2.32 ± 0.13) were significantly higher in the strip sites when compared to both the 2020 control (GLMM, A: 17.87 ± 5.07 ; D: 1.64 ± 0.1 , $df=1$, $z=$ A: 5.23; D: 6.28, respectively, $p < 0.001$) and 2020 natural treatments (GLMM, A: 6.3 ± 1.48 ; D: 1.32 ± 0.073 , $df=1$, $z=$ A: 4.36 ; D: 9.06, respectively, $p < 0.001$). In 2021, abundance (GLMM, 9.67 ± 1.25 , $df=1$, $z=7.55$, $p < 0.001$) and diversity (GLMM, 1.50 ± 0.06 , $df=1$, $z=7.78$, $p < 0.0001$) of carabids at strip sites declined compared to the previous year. In 2021, carabid abundance and diversity were similar between strip and control sites (A: 17.5 ± 3.78 ; D: 1.46 ± 0.06), but both treatments collected more carabids than natural sites (2.1 ± 0.46).

Across all sampling methods, we did not find a difference in bee abundance or diversity among treatments (Figure 4.2C, D, Table 4.2). Strip sites contained the most

bees overall (4.66 ± 0.62) and contained roughly twice as many bees as control (2.01 ± 0.26) and natural sites (2.20 ± 0.25). Both overall bee abundance ($+0.32 \pm 0.23$ bees per year) and diversity ($+0.58 \pm 0.09$ per year) increased as the strips established. Across all sampling methods, we did not find a difference in syrphid abundance (0.67 ± 0.08) or diversity (1.074 ± 0.007) between treatments (Figure 4.2E, F, Table 4.2)

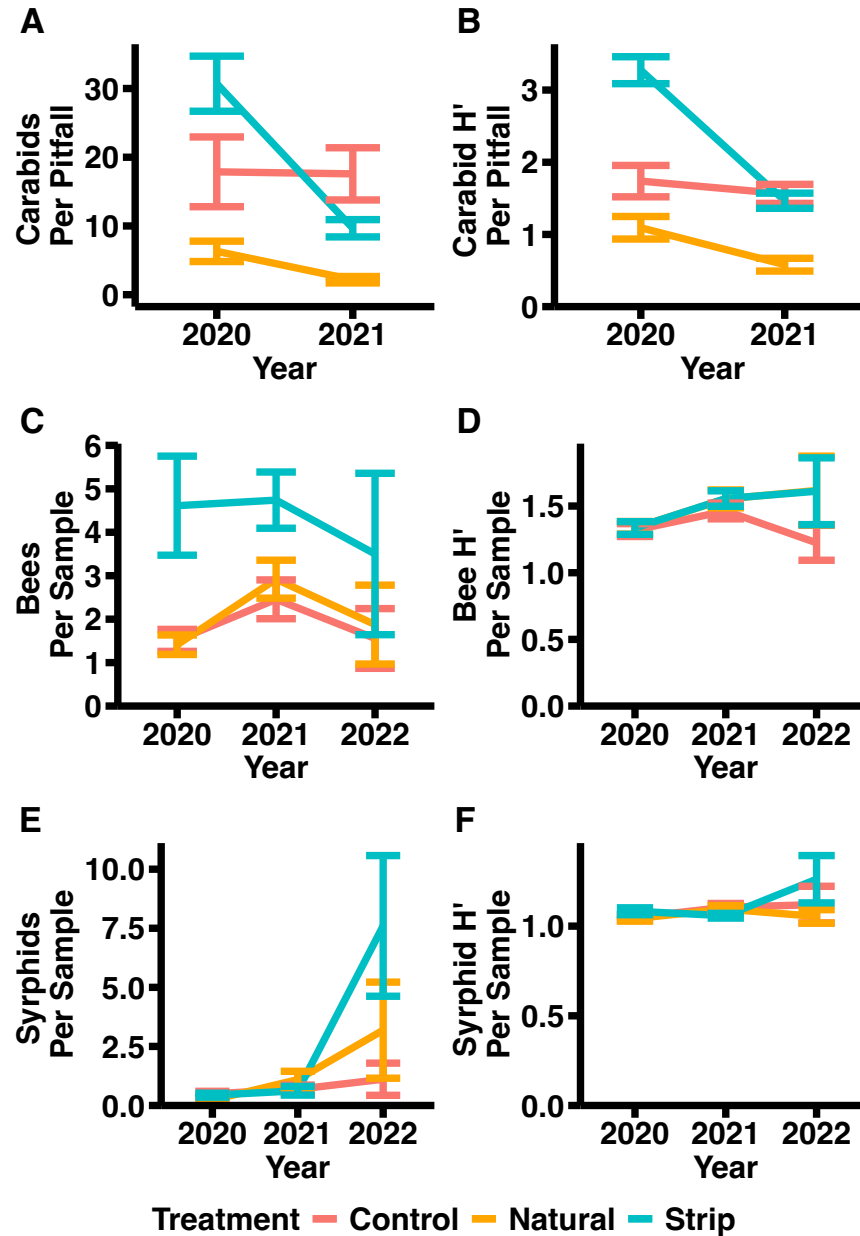


Figure 4.2 Average abundance (A, C, E) and Shannon diversity (H': B, D, F) of ground beetles per pitfall trap (A, B) and both bees (C, D) and syrphids (E, F) across three sampling methods. Data were collected in the summer of 2020–2022 in Manitoba, Canada. Treatments included strip, locations adjacent to crops with floral strip; control, locations adjacent to crops without any enhancements; and natural, semi-natural areas without management. Sampling methods included: carabids: pitfall traps; bees and syrphids: blue vane traps, pan traps, and net transects. Figure shows mean $\pm$ SE of values. Figure was created using ggplot in R (R Core Team, 2023; Wickham, 2016). Sample sizes (A, B) strip= 43; control= 43; natural= 34, (C, D, E, F) strip= 69; control= 69; natural= 56.

Table 4.2 Statistical results from generalized linear mixed effect models (GLMM) comparing insect abundance and Shannon diversity across sites of three treatments. Data were collected at sites in Manitoba, Canada in the summers of 2020–2022. Treatments included strip, locations adjacent to crops with floral strips; control, locations adjacent to crops without any enhancements; and semi-natural, semi-natural areas without management. All models included site as a random effect and an offset to standardize active collection hours for pan traps.

Taxon	method	measure	treatment			year			treatment * year		
			df	X ²	<i>p</i>	df	X ²	<i>p</i>	df	X ²	<i>p</i>
Carabidae	pitfall	abundance	2, 553	31.74	< 0.0001	1, 554	0.14	0.70	2, 553	22.2	< 0.0001
bees	all		2, 2998	3.37	0.19	1, 2999	5.43	0.002	2, 2998	1.51	0.47
Syrphidae	all		2, 2998	5.61	0.06	1, 2999	4.06	0.04	2, 2998	5.65	0.06
Carabidae	pitfall	diversity	2, 553	42.28	< 0.0001	1, 554	2.17	0.14	2, 553	25.08	< 0.0001
bees	all		2, 2998	0.29	0.86	1, 2999	26.4	< 0.0001	2, 2998	0.47	0.79
Syrphidae	all		2, 2998	2.84	0.24	1, 2999	89.8	< 0.0001	2, 2998	2.56	0.28

4.4.4 Beneficial insect spillover

There was evidence of bee spillover into crops during canola bloom and all crops combined, but there was no evidence of spillover effects for other groups in either field type. Within the field edge, the strip sites contained significantly more carabids (17.5 ± 1.82) than the OSC (GLMM, 12.5 ± 2.6 , $df=1$, $z=6.81$, $p<0.0001$, Figure 4.3A).

However, there were no significant differences in carabid abundance within the fields and no differences in carabid Shannon diversity (Figure 4.3B) were found. Bees collected within non-blooming fields were significantly more abundant and diverse (Figure 4.3C, D) at strips (A: 0.25 ± 0.016 ; H': 1.05 ± 0.0007) when compared to OSC sites (A: 0.20 ± 0.013 ; H': 0.03 ± 0.005) regardless of distance within the field (Table 4.3). We found similar trends when we analyzed blooming canola fields as bees were more abundant within the strip (0.79 ± 0.18 per sample) treatment than the OSC (0.39 ± 0.05), regardless of distance. However, overall bee diversity from blooming fields (1.08 ± 0.01) did not change between treatments or distance (Table 4.3).

The abundance of syrphids collected within blooming and non-blooming fields (Figure 4.3E) was not significantly different between treatments or distance (Table 4.3). Syrphid diversity from non-blooming fields was higher in the OSC (1.02 ± 0.003) than the strip treatment (0.01 ± 0.002 , Figure 4.3F), but diversity from blooming fields was similar between treatments. Syrphid diversity at either field type did not change as distance increased (Table 4.3).

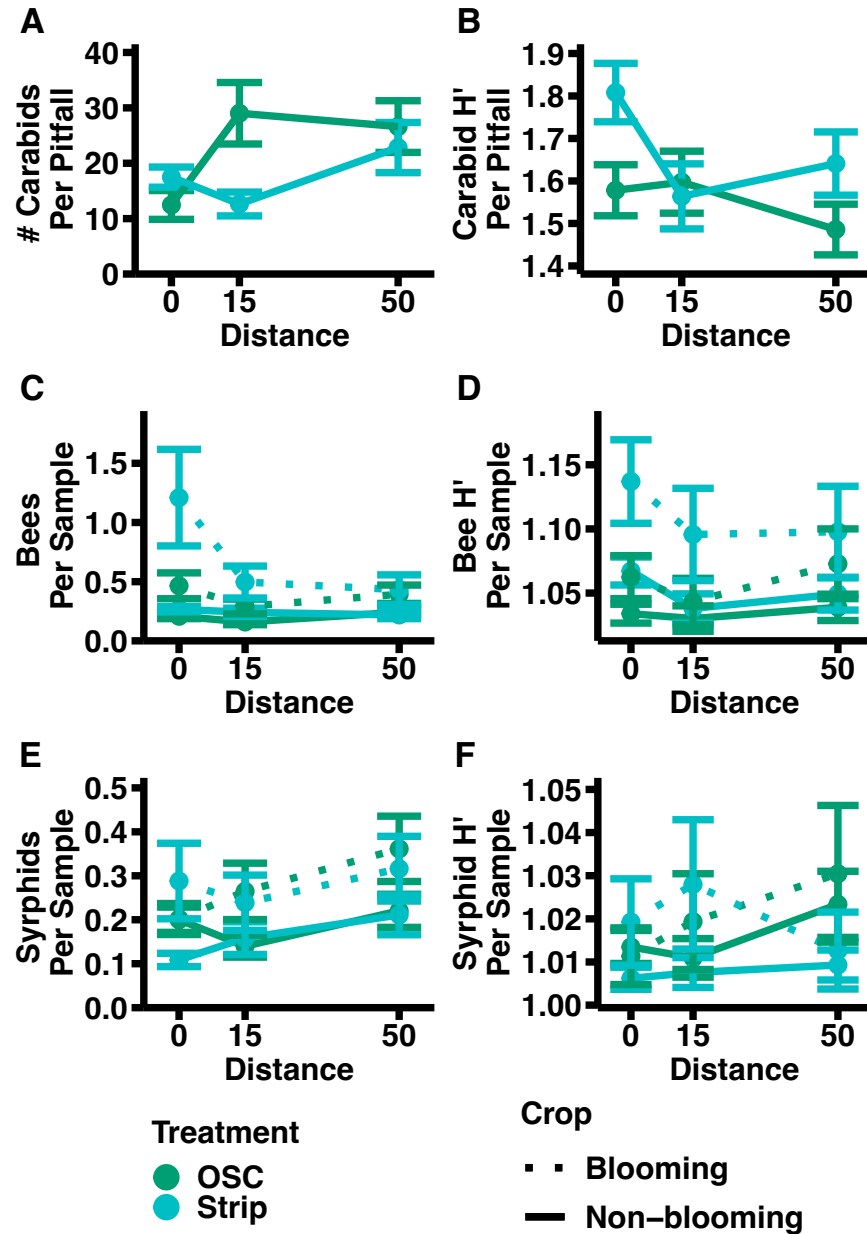


Figure 4.3 Average ground beetle abundance (A) and Shannon diversity (H') (B) per pitfall trap, average bee abundance (C) and H' (D) per sample, and average syrphid abundance (E) and H' (F) per sample. Abundance and diversity measures were collected at strip and OSC edges and up to 50 meters into fields in front of both treatments. Bee (C, D) and syrphid (E, F) blooming crop samples included both pan trap and net transects while non-blooming samples included only pan traps. Data were collected in the summer of 2020–2022 in Manitoba, Canada. Treatments included strip, locations adjacent to crops with floral strip; and OSC, a separate, unmanaged, edge of the strip field that was unmanaged. OSC sampling locations were at least 150 m away from edges of floral strips. Figure shows mean $\pm$ SE of abundance and H' values. Figure was created using ggplot in R (R Core Team, 2023; Wickham, 2016). Sample sizes (A, B) strip, 43; OSC, 43; (C, D, E, F) non-blooming strip, 59; non-blooming OSC, 58; blooming strip, 17; blooming OSC, 17.

Table 4.3 Statistical results from generalized linear mixed effect models (GLMM) comparing insect abundance and diversity between strip and OSC edges and within the fields up to 50 m. Data were collected at sites in Manitoba, Canada in the summers of 2020–2022. Treatments included strip, locations adjacent to crops with floral strips and OSC, locations adjacent to crops without any enhancements but was located at the same field as the strip. Distance into the field included 15 and 50 m away from either the strip or OSC edge. All models included site as a random effect and an offset to standardize active collection time for pan traps.

Taxon	method	crop	measure	treatment			distance			treatment * distance		
				df	X ²	<i>p</i>	df	X ²	<i>p</i>	df	X ²	<i>p</i>
Carabidae	pitfall	all	abundance	1, 925	26.32	< 0.0001	2, 925	13.45	0.0002	2, 925	18.67	< 0.0001
bees	pan trap			1, 3794	9.14	0.003	1, 3794	1.82	0.18	1, 3794	3.28	0.07
Syrphidae	pan trap			1, 3794	1.37	0.24	1, 3794	0.89	0.35	1, 3794	0.0002	0.99
bees	all	blooming canola		1, 1103	13.16	0.0002	1, 1103	1.78	0.18	1, 1103	1.3	0.25
Syrphidae	all			1, 1103	1.59	0.21	1, 1103	1.83	0.18	1, 1103	0.31	0.57
Carabidae	pitfall	all	diversity	1, 925	3.42	0.06	2, 925	1.83	0.17	2, 925	0.04	0.85
bees	pan trap			1, 3794	8.52	0.003	1, 3794	0.05	0.83	1, 3794	0.12	0.73
Syrphidae	pan trap			1, 3794	5.06	0.024	1, 3794	0.09	0.75	1, 3794	0.04	0.85
bees	all	blooming canola		1, 1103	1.93	0.16	1, 1103	1.61	0.2	1, 1103	0.28	0.59
Syrphidae	all			1, 1103	0.05	0.82	1, 1103	2.74	0.09	1, 1103	0.13	0.72

4.4.5 Local and landscape effects on insect abundance and diversity

Carabids and bees both responded to local scale variables, but syrphid flies did not (Table 4.4). The strongest landscape effects varied between our three beneficial insects. Carabid capture rates were higher in areas with less ground cover of grass and forbs and less habitat fragmentation (edge density) in the immediate landscape (500 m). Bee capture rates were higher in areas with increased forb cover, higher Shannon landscape diversity, and less developed lands within the broad surrounding areas (3 km). Bee diversity was influenced at a smaller (1 km) spatial scale, though they were not significantly associated with any variables tested here. Both syrphid abundance and diversity was negatively associated with edge density and abundance was positively associated with landscape diversity.

Table 4.4 Local and landscape level effects on beneficial insect abundance (A) and Shannon diversity (H') that were included in the final generalized linear mixed effect models. Beneficial insects were collected in the summers of 2020–2022 in Manitoba Canada across sites of three treatments. Treatments included strip, locations adjacent to crops with floral strips; control, locations adjacent to crops without any enhancements; and semi-natural, semi-natural areas without management. Model terms included insect abundance (A), insect Shannon diversity (H'), treatment (T), year (Y), numerical day of year (DOY), offset for pan trap sampling hours (HRS), average wind speed (AW), percent bare ground cover (BA), blooming forb Shannon diversity (BH'), blooming forb richness (BR), percent developed landcover (DV), landscape edge density (ED), percent forb cover (FB), percent grass cover (GR), landscape Shannon diversity (LH'), maximum wind speed (MW), percent natural landcover (NAT), percent non-rewarding agriculture landcover (NRA), and percent rewarding agriculture landcover (RA). Significance of variables is shown adjacent to model terms or beside effect sizes: * $p < 0.05$; ** $p < 0.01$; *** $p < 0.001$.

Taxa	Collection method	Spatial scale	Model	df	VIF
Carabidae	pitfall trap	local	A ~ T* + Y (-0.02) + TxY*** + DOY (-<0.01) + GR (-0.61***)	500	T (723.3), Y (3.23), TxY (651.7), DOY (1.57), GR (3.23)
			H' ~ T*** + Y (-0.15***) + TxY*** + DOY (-0.10*) + FB (-0.17**)	500	T (953.72), Y (3.54), TxY (774.6), DOY (1.61), FB (1.99)
		0.5 K	A ~ T*** + Y (-0.01) + TxY*** + DOY (-0.08) + ED (-0.83***)	545	T (596.2), Y (3.4), TxY (615.4), DOY (1.62), ED (1.83)
			H' ~ T*** + Y (-0.16*) + TxY*** + DOY (-0.13**) + ED (-0.20***)	545	T (807.8), Y (3.68), TxY (862.7), DOY (1.55), ED (1.62)
		local	A ~ T + Y (0.64***) + TxY + HRS + DOY (0.77***) + FB (0.28*)	2970	T (798.3), Y (3.28), TxY (791.6), DOY (1.25), FB (1.71)
			H' ~ T + Y (0.24***) + TxY + HRS + DOY (0.14***)	2991	T (591.8), Y (3.06), TxY (644.5), DOY (1.02)
bees	all	3 K	A ~ T* + Y (0.53***) + TxY + HRS + DOY (0.74***) + LH' (0.26***) + DV (-0.34***)	2989	T (774.2), Y (3.67), TxY (831.1), DOY (1.23), LH' (1.04), DV (1.13)
		1 K	H' ~ T + Y (0.24***) + TxY + HRS + DOY (0.14***)	2991	T (591.8), Y (3.06), TxY (644.5), DOY (1.02)
		local	A ~ T* + Y (0.46**) + TxY + HRS + DOY (0.30***)	2991	T (495.6), Y (3.0), TxY (538.2), DOY (1.12)
Syrphidae	all	local	H' ~ T + Y (0.70***) + TxY + HRS + DOY (0.24***)	2991	T (527.9), Y (3.06), TxY (573.8), DOY (1.02)
			A ~ T* + Y (0.52**) + TxY* + HRS + DOY (0.28**) + LH' (0.72***) + ED (-1.36***) + DV (0.35**)	2988	T (617.4), Y (3.43), TxY (692.01), DOY (1.23), LH' (2.52), ED (3.2), DV (1.44)
		0.5 K	H' ~ T + Y (0.70***) + TxY + HRS + DOY (0.24***) + ED (-0.14*)	2990	T (535.2), Y (3.07), TxY (580.1), DOY (1.2), ED (1.57)

4.4.6 Insect community assemblages

Communities of all three beneficial insects differed between both treatment and years as there was an interaction between these two variables (Table 4.5). Beetle communities were all similar in 2020 but in 2021 strip and control communities were unique (PERMANOVA, $df=1$, $F=4.51$, $p=0.03$, Table S 7.6). Bee communities collected from strip sites tended to be unique from other treatments but slowly became similar to other treatments by 2022 (Figure S 7.7, Table S 7.6). Syrphids responded similarly to bees, but control sites tended to be the unique community of syrphids instead of strip sites (Figure S 7.8, Table S 7.6).

Edges and field samples for bees and syrphids tended to be unique at all four sampling locations (OSC, strip, and field samples for both treatments, Table 4.6). However, communities of ground beetles were similar as a pairwise test showed no significant differences between sampling locations (Figure S 7.9). Bee communities were different between edges and within field samples for strip (PERMANOVA, $df=1$, $F=4.70$, $p=0.004$) and OSC treatments (PERMANOVA, $df=1$, $F=2.67$, $p=0.018$, Figure S 7.10, Figure S 7.11). Additionally, OSC and strip edges (PERMANOVA, $df=1$, $F=6.83$, $p=0.003$) and OSC and strip field samples (PERMANOVA, $df=1$, $F=2.24$, $p=0.034$) were different. Syrphid communities were different between edges and within field samples for strip treatments (PERMANOVA, $df=1$, $F=6.85$, $p=0.006$), while OSC edge and field communities were similar (Figure S 7.12, Figure S 7.13). Additionally, OSC and strip edges (PERMANOVA, $df=1$, $F=25.13$, $p=0.001$) and field samples (PERMANOVA, $df=1$, $F=15.02$, $p=0.001$) were different.

Table 4.5 Statistical results from permutational analysis of variance tests of insect communities collected at edges of sites of three treatments in Manitoba, Canada in the summers of 2020–2022. Treatments included strip, locations adjacent to crops with floral strips; control, locations adjacent to crops without any enhancements; and semi-natural, semi-natural areas without management.

Taxon	method	Treatment			Year			Treatment * Year		
		df	F	<i>p</i>	df	F	<i>p</i>	df	F	<i>p</i>
Carabidae	pitfall	2, 552	3.40	0.001	1, 553	1.42	0.34	3, 552	1.67	0.004
bees	all	2, 2997	24.2	0.001	1, 2998	2.11	0.48	2, 2997	2.24	0.006
Syrphidae	all	2, 2997	6.35	0.003	1, 2998	4.40	0.29	2, 2997	2.85	0.004

Table 4.6 Statistical results from permutational analysis of variance tests of insect communities collected at strip fields comparing the strip and OSC treatments. Data were collected at sites in Manitoba, Canada in the summers of 2020–2022. Treatments included strip, locations adjacent to crops with floral strips and OSC, locations adjacent to crops without any enhancements but was located at the same field as the strip. Distance into the field included 15 and 50 m away from either the strip or OSC edge. Edge vs field comparisons included the edges (strip edge or OSC edge) and within field samples (strip field or OSC field).

Taxon	method	crop	treatment			distance			treatment * distance			edge vs field		
			df	F	<i>p</i>	df	F	<i>p</i>	df	F	<i>p</i>	df	F	<i>p</i>
Carabidae	pitfall	all	1, 925	2.38	0.01	2, 925	0.89	0.29	2, 925	0.94	0.23	3, 927	1.69	0.014
bees	pan trap		1, 3795	6.07	0.001	2, 3795	2.38	0.001	2, 3795	3.7	0.001	3, 3797	4.63	0.001
Syrphidae	pan trap		1, 3795	4.25	0.141	2, 3795	0.78	0.13	2, 3795	1.43	0.012	3, 3797	2.59	0.014
bees	all	blooming canola	1, 1100	5.34	0.001	2, 1100	2.65	0.001	2, 1100	2.89	0.001	3, 1102	4.26	0.001
Syrphidae	all		1, 1100	37.36	0.001	2, 1100	2.18	0.011	2, 1100	3.06	0.003	3, 1102	13.73	0.001

4.5 DISCUSSION

4.5.1 Strip establishment

Overall, floral strips significantly increased forb cover and blooming forb richness when compared to control and natural sites. The vegetation structure of the strips changed over time as grass cover increased and forb cover decreased. Regardless of this, strips contained the most forb cover of any treatment across all sampling years. We also noted less bare ground cover in 2022, a sign that the vegetation within the floral strips was expanding as it established. Annuals were used in the seed mix (Table S1) to provide ample ground cover to suppress weeds in the first year of establishment and to hopefully provide a suitable area for the perennials to establish. Thus, we expected plant structure to change as strips establish and it appears that it continues to change for at least the first three years. The vegetation structure may be close to equilibrium since bare ground cover was lowest at strip sites in 2022 (Table 1). Other studies have found that it may take up to four years for strips to produce measurable effects on the surrounding crop and insect communities (Blaauw and Isaacs, 2014). The strips should be monitored for subsequent years to track long term establishment of the vegetation community and to determine the length of time necessary for vegetation community to stabilize.

Extreme flooding and drought conditions made strip establishment more difficult than expected. Some details are discussed in our previous study (chapter 3) regarding the effects of strips during their first year of establishment (Killewald et al., 2024) and we

continued to have extreme weather variations for almost every year of sampling for this study. In 2020, Manitoba received fairly normal amounts of precipitation (Hydrologic Forecast Centre 2020). However, the following years provided both drought (Hydrologic Forecast Centre 2021) and severe flooding (Hydrologic Forecast Centre 2022) conditions resulting in major wildfires and flooding across the province. In 2022, at least one strip was submerged under 10 cm of water for part of the growing season due to the amount of precipitation. Regardless of extreme conditions, strips established well and contained the most forb cover and blooming forb richness of any treatment. Additionally, we found that the strips positively affected the beneficial insect communities found within them, but these responses varied by insect group.

4.5.2 Carabidae

Both capture rates and Shannon genus diversity of ground beetles showed a significant interaction between treatment and year, where strip sites in 2020 captured twice as many beetles with higher diversity per pitfall trap than control sites. However, carabid capture rates and diversity declined in 2021 to similar levels as control sites. Although pitfall traps are one of the most common ways to capture carabids (Bater 1996), they measure the activity density of ground beetles and do not directly measure population size. This means that beetles can be captured more frequently because they are either more active, more abundant, or both (C. F. G. Thomas et al., 1998).

Ground beetle capture rates can also be influenced by density of local vegetation (Greenslade, 1964; Hohbein & Conway, 2018) since dense vegetation makes it harder for carabids to translocate. Greenslade (1964) noted that beetles can move more easily

through areas of sparse vegetation, thus making them more likely to be captured by the pitfall traps. Here and previously (Killewald et al., 2024), we found that ground beetles were negatively associated with the percent grass cover at the local scale. Strip sites showed decreasing amounts of forbs and increasing amounts of grasses as they established, so it is possible that the declines in carabid capture rates could be caused by the increased stem density of grasses in later years of sampling within the strips. Ground beetle capture rates were higher in strips than control fields for at least the first (Killewald et al., 2024) and second years of strip establishment, but we cannot determine if their capture rates were higher due to vegetation structure or due to a numerical response from the carabids. Although not implemented here, future studies should create a standardized patch of bare vegetation (Greenslade, 1964) across treatments to remove vegetation density as a variable and more accurately compare ground beetle capture rates.

At the landscape scale, we found a negative association between edge density at 500 m and both carabid capture rates and diversity, indicating that at the relatively small scale, habitat fragmentation may reduce ground beetle capture rates. We found opposite patterns during our first year of this study, as species level beetle diversity in 2019 was positively associated with edge density at 0.5 km (Killewald et al., 2024). Many carabid species are incapable of flight or cannot fly far (Lindroth, 1969a, 1969b, 1968, 1966, 1963, 1961) so it is possible that increasing edge density makes it harder for ground dwelling beetles to translocate. It may be more difficult to move long distances if the chances of intercepting a hostile landcover type are increased. Thus, floral strips may aid in beetle movement as they could provide a ‘beetle highway’ that can be used to move

past possibly hostile landcover types- though this needs to be tested further to verify this hypothesis.

In 2019 (Killewald et al., 2024), beetle abundance was more evenly spread across genera, as we found five genera that made up >10% of total samples. However, in 2020 onwards, only two genera, *Pterostichus* and *Harpalus*, made up >10% of total captures. During the first year of the study, the strips were likely inhabited with highly mobile species that could colonize the new habitat while communities in 2020 onwards likely included fast reproducing species (Holliday et al. 2014). Ground beetles from this study would have to be identified to species to confirm this theory. It appears that strip sites are able to house a representative ‘natural’ community of ground beetles than those found within control fields as the communities between the strip and the natural treatments were similar while the strip and control differed. Ground beetle communities in 2020 and 2021 were different between treatments, but not years, indicating that communities at each of the treatments are relatively stable. Beetles collected within the fields were also similar to the beetles collected on the edges of the fields, indicating that increases in beetle abundance at strip sites were caused by local beetle populations rather than those coming from separate populations in the landscape. In our previous study (Killewald et al., 2024) we found that the strips contained a unique community of beetle species when compared to all other treatments. Thus, it seems that strip communities changed between 2019 and 2020 and moved towards a more ‘natural’ community than an agriculturally dominated one. Although identifying beetles to species here would have been a more accurate way to compare communities, we could only identify them to genus due to funding and time constraints. These samples are stored at the Wallis-Roughley Museum

of Entomology at the University of Manitoba, and it is our hope that these samples may be identified to species in the future to more accurately compare beetle community structure. If possible, we recommend that future studies identify specimens to species for community-based analysis.

4.5.3 Bees

Though we found no statistical difference in bee abundance or diversity between treatments, overall bee abundance was higher in strip sites when compared to control or natural sites. The first year of this project found significantly higher abundances of bees collected with nets in strip sites (Killewald et al., 2024) and is supported by previous studies as well in the United States , the U.K. (Campbell et al., 2017), and Canada (Van Drunen et al. 2022). Bee abundance and diversity in our study both increased as the floral strips established, indicating that the strips can support long-term populations of bees. We hope that bee abundances will continue to increase as strips establish and continued monitoring at these sites should be in place to verify these trends.

Bees responded positively to the percent forb cover at the local scale. These results are unsurprising given that a lack of floral resources has been a major driver of bee declines (Kremen et al. 2007) and the average percent of forb cover identified from non-enhanced areas in our study was particularly low (Table 1). Bee abundance responded strongest at the largest, 3 km, spatial scale while diversity responded strongest at 1 km. This indicates that overall population density is regulated at the broader landscape scale while diversity may be regulated more so within the immediate

landscape. Other studies have found that the same landscape variable (habitat fragmentation) can be both positively and negatively related to bee population metrics depending on the spatial scale (Clake et al., 2022).

Bee abundance was positively associated with landscape Shannon diversity and negatively associated with developed lands within 3 km. Other studies have found that bees respond positively to increased quality landcover (Kennedy et al. 2013) and increasing landscape diversity increases the chance of bees finding appropriate floral resources. Natural or unmanaged land within the agricultural landscape of Manitoba is scarce (Agriculture and Agri-Food Canada, 2021) so installing floral strips can help to increase not only local resources but overall landscape complexity. Both factors that have been identified here to support bee population density and diversity.

In 2020, strip sites contained a distinct community of bees compared with all other treatments. By 2022, bee communities in strip sites were similar to other treatments. This indicates that strips support the entire community of bees and not just a subset of species. Differences in community composition at first could be caused by colonizing species as other, possibly more specialist, species may take longer to establish. Nonetheless, the increases in bee abundance coupled with the finding that bee communities at strips match the surrounding landscape after three years of establishment strongly support the use of strips in agricultural production. Though we did not find an overall effect of treatment on bee abundance or diversity, Blaauw and Isaacs (2014) found that it may take four years to find measurable effects from the wildflower plantings. Since rotational cropping systems are more unpredictable than perennial systems, it may take more time for beneficial insect communities to fully establish within

the floral strips. Thus, continued monitoring at these sites should be in place to verify community trends.

4.5.4 Syrphidae

Strips were not effective at increasing the overall abundance or diversity of syrphids as we found no significant differences between treatments- a finding that is supported by our previous study as well (Killewald et al., 2024). Other studies have reported higher syrphid abundance and diversity from strips when compared to control sites (Blaauw and Isaacs, 2014; Haenke et al., 2009) and it still remains unclear why syrphids show different responses in our study. Most syrphid captures from pan traps were from one species, *Toxomerus marginatus* (Say, 1823, 77.2% of captures). This species is common across North America (Skevington et al. 2019) and it has been the most common syrphid species from other studies in Manitoba as well (Killewald et al., 2024; Samaranayake & Costamagna, 2019). It is likely that the syrphid community in our study is different from that of other studies performed in different regions and as such, may show different responses. Nonetheless, it appears that strips are not effective at increasing syrphid abundance or diversity on Manitoba farm land. The *Toxomerus* species identified here are aphidagous as larvae (Hopper et al. 2011), so their population sizes are likely influenced by prey availability during larval stages rather than floral resources available to adults.

The abundance and diversity of syrphids is influenced mostly at the local scale (< 0.5 km) where abundance is increased in areas with larger patches of different landcover types. Thus, our results indicate that areas with fields of multiple types of crops should

increase syrphid abundance and diversity. Other studies have found that syrphid abundance and diversity increases with increased landscape complexity (Ortega et al. 2023). Though we found no effect of the strips on syrphid communities, installing strips or diversifying agricultural practices can increase the landscape diversity that helps to support beneficial insect populations (Aguilera et al. 2020). Syrphid communities at strip sites were similar to other treatments more often than not. In most cases, the control fields contained the distinct treatment of syrphids while strip and natural sites were similar. Although strips did not increase syrphid abundance or diversity, they did not appear to change the long-term community structure of syrphids and likely do not have any negative effects on their populations- further testing is needed to verify long-term trends.

4.5.5 How do strips compare to natural sites?

Unmanaged habitats should provide better overwintering habitats for beneficial insects (Griffiths et al. 2008) than crop fields with regular disturbances. Thus, we expected that the natural sites would be similar to strip fields due to their low amounts of disturbance. Even though we found that natural sites contained an intermediate amount of total forb cover between control and strip sites, the amount of blooming forb cover at natural sites was similar to control fields. Similarities in blooming vegetation structure may explain why bees were collected in similar abundances between natural and other sites. However, the results for ground beetles are interesting as natural sites contained less ground beetle captures and diversity than other treatments. We found similar communities of ground beetles between the natural and most other treatments, indicating

that natural sites support similar community structure of ground beetles, but at a possibly lower abundance. Many of the species of ground beetles captured here are associated with agricultural landscapes (Holliday et al. 2014), which may be because the agricultural landscape dominates southern Manitoba (Agriculture and Agri-Food Canada, 2021). Natural sites contained less grass cover than control sites, so vegetation density does not appear to have caused the lower capture rates at natural sites. Thus, it appears that natural sites may not be as hospitable to ground beetles in our study as agriculture fields. This could be caused by an overabundance of agriculturally associated species. However, beetle samples should be identified to species to determine species specific responses as these may vary based on ecological traits.

Given the lower capture rates of ground beetles and the similar capture rates of bees at natural and control fields, it seems that the seminatural sites in Manitoba are not of high enough quality to enhance beneficial abundances by themselves. The amount of high-quality habitats in the landscape has been positively associated with beneficial insects (Kennedy et al., 2013; Purtauf et al., 2005), which is generally provided by natural sites. Insects rely on undisturbed sites as agricultural disturbances have resulted in a loss of almost half the number of individuals and a quarter of the insect species in the surrounding areas (Outhwaite et al., 2022). This amplifies the need to install these strips in Manitoba especially since almost 50% of the available land south of the 51st parallel is used for farming (Agriculture and Agri-Food Canada, 2021). It seems clear that installing floral strips increases beneficial insect abundance and diversity when compared to natural sites, so land managers should consider installing strips instead of simply leaving land unmanaged for long periods of time.

4.5.6 Do floral strips increase beneficial insect spillover?

Previous studies suggest that beneficial insects in strips will spill over into adjacent habitats (Morandin et al., 2014) and we expected to find an increase in beneficial insect abundance within adjacent fields. Insect spillover in our study varied between insect groups. Capture rates of ground beetles and syrphids were not different within the fields, but we did see an increase in bee abundances within fields in the strip treatment. This effect was identified from both blooming and non-blooming crops, so it appears that strips are effective at increasing bee abundances and that this effect spills over at least 50 m into the adjacent crop fields. However, bee diversity was not increased within the fields, so it appears only select species may spill over into adjacent crops. *Apis mellifera* was the most commonly collected bee on field edges. However, it does not appear that this species spills over into the crops as two wild bee species, *Lasioglossum albipenne* (Robertson) and *Lasioglossum semicaeruleum* (Cockerell), were the most commonly collected bees from within fields. Thus, it appears that spillover of wild bees into adjacent crops may be more common than spillover of honey bees.

Increases in beneficial insects within the fields can increase ecosystem services such as pollination (Blaauw and Isaacs, 2014) and biological control (Blaauw and Isaacs, 2015) which can both increase yield (Woodcock et al. 2019). Some studies find that wild bees are more efficient pollinators of some crops than honey bees (Garibaldi et al. 2013), so increases of wild bee abundance within fields in our study is promising. Although not directly tested here, we hope that the increases in bee abundance within blooming canola crops will result in increases in pollination and higher yields. These trends have been

identified from canola crops in Alberta, Canada (Morandin & Winston, 2005) so our results are promising.

4.6 CONCLUSIONS

It is clear that agricultural intensification and land use change have altered long-term insect community structure (Outhwaite et al., 2022; Raven & Wagner, 2021; Wagner, 2020). Stronger interventions are needed as those already in place still result in great losses of insects in agricultural areas (Mancini et al. 2023). The use of floral strips has shown to be effective for local insect management (Ganser et al., 2021; Haaland et al., 2011) and were tested here for their effects on beneficial insects in rotationally managed crop systems. Floral strips established well, even during extreme weather conditions, and significantly increased both forb cover and blooming forb richness when compared to other treatments. In response to this, we found increased capture rates and Shannon diversity of ground beetles but syrphids did not respond to the addition of strips. Though we did not identify an overall effect of treatment on bee capture rates, their abundance was highest in strips across all sampling years, indicating that they benefit from the addition of floral resources. Bee abundance was significantly higher within adjacent fields (both all field types and blooming canola) indicating that the floral strips increase spillover of bees in multiple crop types. Both bee abundance and diversity continued to increase as strips established, indicating that strips are effective at maintaining populations of bees long-term. Natural site comparisons were not as effective at supporting beneficial insect populations as strips, so land managers should consider installing floral enhancements instead of leaving land unmanaged for long

periods of time. Policy changes aim to support insect communities, but farmer adoption of insect friendly practices is low and most farmers would only adopt insect-friendly measures if economically viable (Busse et al. 2021). Additional efforts should be implemented to increase support to protect valuable resources, install additional resources, and or to diversify farming systems (Hufnagel et al., 2020). Our results here show that floral strips are an effective management tool for multiple groups of insects and may increase the ecosystem services that they provide to agriculture.

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CHAPTER 5. FLORAL STRIPS ADJACENT TO ROTATIONALLY MANAGED CROP FIELDS SIGNIFICANTLY INCREASE NESTING DENSITY AND SUPPORT POLLEN FORAGING OF LEAFCUTTER BEES

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5.1 ABSTRACT

1. Bees in many regions of the world are in decline increasing evidence that agricultural practices are to blame. Although wildflower enhancements adjacent to croplands have proven effective at increasing wild bee abundance and diversity, how these enhancements can improve the diet and nesting rates of bees remains poorly known.
2. We placed hollow reeds next to floral strips, control fields, and semi-natural areas in Manitoba from 2019–2021 to determine how pollen utilization and provisioning rates of stem-nesting leafcutter bees (*Megachile*) differed between treatments.
3. We found that strip sites significantly increased nesting density of *Megachile* when compared to the other treatments. Strip sites contained the most pollen species, including 30% of species sown into the floral strip itself, suggesting that *Megachile* use the strip for pollen provisioning.
4. At the landscape scale, nesting density of *Megachile* responded positively to the amount of rewarding agriculture (crops frequently visited by insects for floral rewards). However, the number of offspring per nest responded negatively to rewarding agriculture and local Shannon vegetation diversity, indicating that *Megachile* create more nests with fewer offspring in areas of large-scale flowering crops or higher vegetation diversity. Possible reasons for this are discussed.
5. Our research supports the idea that floral strips increase beneficial insect abundance and that floral strips supplement pollen foraging of bees.

Keywords

Conservation; Megachile; Pollination; Wildflower Strips

5.2 INTRODUCTION

Bees across the globe are in decline (Cameron et al., 2011; Sánchez-Bayo & Wyckhuys, 2019) with increasing evidence of human-induced effects (Lundin et al., 2015; Zattara & Aizen, 2019). Agricultural intensification (Seibold et al. 2019), pesticides (Desneux et al., 2007; Whitehorn et al., 2012), reduction of floral resources (Sánchez-Bayo & Wyckhuys, 2019), diseases (Winter et al. 2006), and many other reasons (Potts et al. 2010) have been identified as major drivers of insect declines. Current agricultural practices promote high yield but often result in a loss of biodiversity and ecosystem services (Stoate et al. 2009). This is problematic since globally 87% of plants (Ollerton et al., 2011) and 35% of food production (Klein et al. 2007) benefit from animal pollination services. Given that population declines of bee species are often caused by multiple factors (Goulson et al. 2015), understanding ways to improve their populations is important for species conservation and securing the future of agricultural production (Kremen et al., 2002).

Floral strips are one way to conserve bee populations. These strips give bees pollen, nectar, and overwintering areas that can increase their abundance and diversity (Haaland, Naisbit and Bersier, 2011; Blaauw and Isaacs, 2014; Williams et al., 2015). Floral strips are often placed in agricultural landscapes since these areas show higher

rates of insect decline (Seibold et al. 2019) and often have fewer natural areas and floral resources available to insects. In areas with large field crops, strips of flowers along the edge of the field are often used (Kordbacheh et al., 2020). These strips can be also shaped as a large rectangular patch near the fields (Blaauw and Isaacs, 2014) or in strips within the fields (MacLeod et al. 2004)—a practice more common in Europe. Since habitat loss from agriculture is one of the main drivers of insect decline (Sánchez-Bayo & Wyckhuys, 2019), these floral strips should be placed in areas where bees need them the most. Floral strips adjacent to agricultural fields have proven effective at increasing bee abundance and diversity in other studies (Haaland, Naisbit and Bersier, 2011; Blaauw and Isaacs, 2014; Williams et al., 2015; Kordbacheh, Liebman and Harris, 2020). However, few studies have documented the effects these enhancements may have on bees' diet (Wood et al., 2017) and reproduction (Ganser et al., 2021; Graham et al., 2020; N. M. Williams et al., 2024).

Pollen nutrition can greatly influence bee population sizes (Brodschneider & Crailsheim, 2010), lifespan (Schmidt et al. 1995), and their ability to fight parasites (Di Pasquale et al. 2013). Nutritional quality of pollen varies between botanical families (Filipiak 2019) and bees often selectively forage on plants to meet nutritional demands (Vaudo et al. 2016). Bumble bees with restricted diets show more evidence of population declines (Wood et al. 2019). Given this, identifying key pollen plants used by different groups of bees is essential for conservation practices. One way to study the pollen preferences of bees is through pollen analysis (Beil et al., 2008). Through removal of scopal loads (Wood et al. 2021) or analyzing pollen provisioned for offspring (Graham et

al., 2020; Killewald et al., 2019), researchers can understand the dietary selection of bee species.

Although most female bees collect pollen, stem-nesting bees are comparatively easy to study since they readily accept artificial nests and can be passively monitored (Krombein, 1967). This gives a less biased view than collecting bees from flowers as samples from trap nests are not limited to the floral species the researcher can locate. Leafcutter bees (Megachilidae: *Megachile*) include important wild and managed pollinator species (Tepedino and Frohlich, 1982; Pitts-Singer and Cane, 2011). *Megachile* accept many types of nesting substrates and species vary greatly in their pollen preferences (O'Neill et al., 2004; Tepedino & Frohlich, 1982). Since females provision only a few cells per day (Cane et al., 1996) and nests can contain over a dozen cells (Stephen & Osgood, 1965), researchers can then identify the pollen use of a female through time. Several studies have examined pollen use of *Megachile* (Cane et al., 1996; Killewald et al., 2019; O'Neill et al., 2004; Stephen & Osgood, 1965; Tepedino & Frohlich, 1982), but few have analyzed how they use pollen from floral strips (Graham et al. 2020). It is unknown to what extent wild *Megachile* will use floral strips for offspring provisioning (Ganser et al., 2021). Identification of pollen loads from *Megachile* nests adjacent to floral strips would also identify key plant species to include in seed mixes for conservation of this group.

Megachile as a group are diverse in both species and global distribution, with over 1,000 species worldwide (Charles Duncan Michener 2007) and 39 species recognized from Canada and Alaska (Sheffield et al., 2011; Gibbs et al., 2023). Of these, several species are important crop pollinators. The alfalfa leafcutter bee, *Megachile*

rotundata (Fab. 1787), is a highly managed and efficient pollinator of alfalfa (*Medicago sativa*, L.) (Pitts-Singer and Cane, 2011) but will also forage on clovers (Richards 2015) and other plants (O'Neill et al. 2004). *Megachile pugnata* (Say 1837) is an Asteraceae specialist and as such, would be useful for sunflower production (Tepedino & Frohlich, 1982). Other members of this genus, such as *Megachile relativa* (Cresson 1878) will collect pollen from a wide range of flowers such as fireweed (*Chamaenerion*), alfalfa, and Asteraceae (Stephen, 1955; Ivanochko, 1979). Many of the described species in Canada will accept trap nests (Sheffield et al. 2011), making them relatively easy to study and perfect model organisms to test how floral strips may change reproductive decisions of bees.

Bees are central place foragers (Sommer et al., 2022) and as such, they are limited to plants within their flight range. Although some bees can fly great distances, most species of *Megachile* will forage within 0.5 kilometres (km) (Greenleaf et al. 2007). Locally, vegetation abundance and diversity (Scheper et al. 2015) influence bee foraging and abundance (Jha & Kremen, 2013). At the landscape scale, agriculture and natural area cover (Kennedy et al. 2013) as well as their juxtaposition has been significantly associated with bee abundance (Clake et al., 2022). In areas of intensive agriculture such as Manitoba, Canada, few natural areas remain in the landscape (Agriculture and Agri-Food Canada, 2021) meaning that foragers likely have few options available. Floral strips may be more effective in areas with few floral resources (Scheper et al. 2015). So analyzing which local and landscape factors best support bees within intensive agricultural areas that may have limited resources is important for long term conservation.

In this study, we used trap nests placed next to floral strips, semi-natural areas, and control fields to compare pollen utilization and nesting rates of *Megachile* across the agricultural region of Manitoba. The objectives of our study are to 1) determine if floral strips increase nesting and offspring provisioning rates of *Megachile*, 2) determine how floral strips and landscape variables influence nesting and pollen collection, 3) compare pollen communities across treatments, and 4) assess if pollen composition changes through time.

5.3 METHODS

5.3.1 Study sites

Forty locations across southern Manitoba were selected for sampling (Figure 5.1). Sites were sampled as part of a project to determine if floral strips adjacent to crop fields can increase beneficial insect abundance (Killewald et al., 2024). These sites contained fields with a floral strip adjacent to one edge (strip, $n=15$), control fields with unenhanced, grassy margins (control; $n=11$), and semi-natural sites ($n=14$). Strip and control fields were rotationally managed crops, whose management was controlled entirely by the producer. Floral strips (Figure 5.2A) were sown with a pollinator friendly seed mix including buckwheat (*Fagopyrum esculentum*, Moench), sainfoin (*Onobrychis viciifolia*, Scop), common sunflower (*Helianthus annuus*, L.), and others (Killewald et al., 2024). Semi-natural sites were locations that were not managed for at least five years

prior to sampling. Most often, these were edges of riparian areas or wildlife management areas.

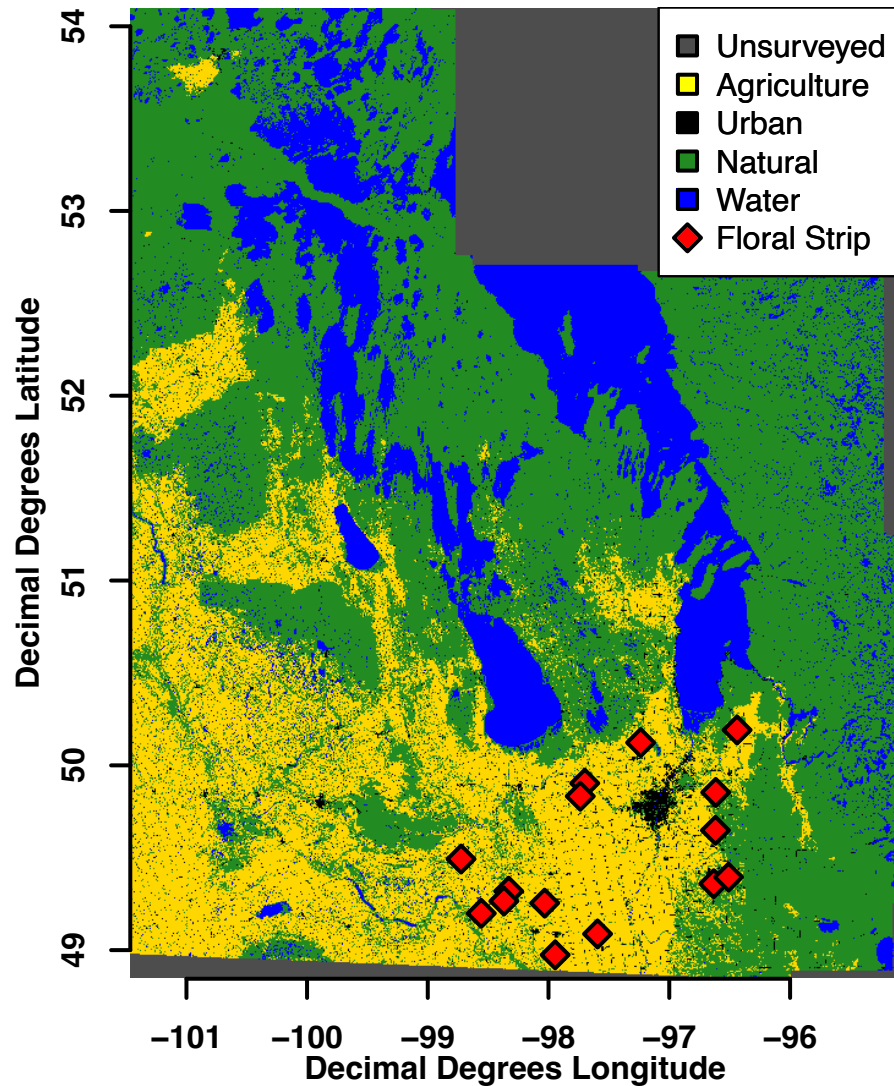


Figure 5.1 Locations of each cluster of sites in Southern Manitoba sampled for nesting *Megachile* during the summers of 2019–2021. Each floral strip had a paired control and semi-natural site located within a 1 km radius. This figure was created using the ggplot2 package in R and data from the 2020 annual crop inventory (Wickham, 2016; Agriculture and Agri-Food Canada, 2021; R Core Team, 2023).

5.3.2 Trap nests

Eighty reeds (*Phragmites*) of various diameters (4–10+ mm) were placed on the edges of sampled sites (Figure 5.2B) within a PVC enclosure to protect from the elements. These trap nests were placed in the fields in the early summer of 2019 and were removed from fields in fall 2021. We used an array of reed sizes since different species of *Megachile* prefer nests with different diameters (Krombein, 1967). Trap nests were checked every time we visited the sites (generally at least once a month during June–October). Any completed nests were removed from the enclosure, replaced with an empty reed of similar size, and then brought back to the lab for processing. Reeds were opened and every other pollen cell was removed for analysis. However, if the larvae had already consumed most of the pollen load, then the cell was not tampered with. The remaining cells were left in the nest, placed into a mesh bag labeled with the collection information, and allowed to diapause in an unheated shed to protect from the elements. We had planned to identify adults from the nests to be able to link pollen collection and nesting density responses with individual species, but unfortunately, dermestids had destroyed most of the nests, so our sample size for this was reduced. Nonetheless, pollen data from the adults that we were able to identify are discussed.

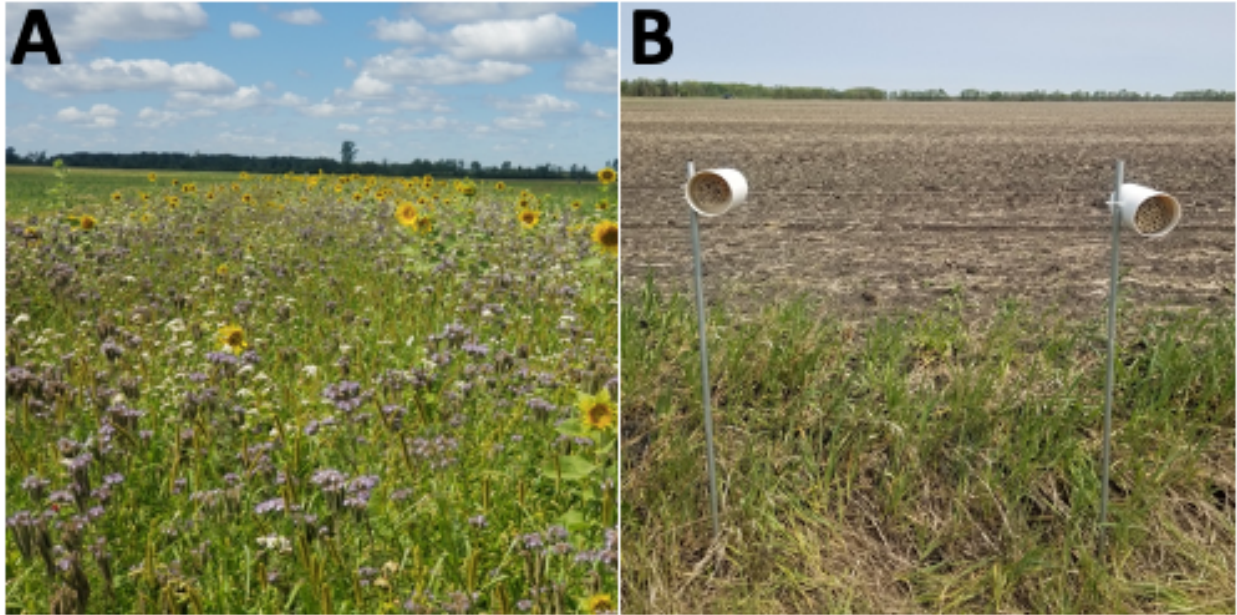


Figure 5.2 Floral strips (A) were contrasted with conventional, grassy field margins (B) and semi-natural sites. Photos were taken in mid-August 2019 (A) or after seeding (B). Trap nests were placed along field edges (B) with and without floral strips or at semi-natural sites.

5.3.3 Local and landscape data

At least once per site per year (mean 1.85 ± 0.09 times per year) we conducted vegetation surveys near the trap nests (Killewald et al., 2024). We placed a 0.5 x 1 m quadrat on the ground at multiple locations along transects and identified the quantity of each blooming plant species within the quadrat. We also recorded the overall percent of forb, grass, and bare ground cover for each quadrat. Afterwards, the quadrats per site per year were averaged together to give an overall vegetation community of the site. The average ground cover of blooming forbs, all forbs (blooming and non-blooming), grasses, and bare ground was used in linear models described below.

Analysis and data extraction were performed using R version 4.3.1 (R Core Team 2023). Landscape data were extracted from the appropriate annual crop inventory available from Agriculture and Agri-Food Canada (Agriculture and Agri-Food Canada, 2021) using the *landscapemetrics*, *raster*, and *sampSurf* packages in R (Gove, Jeffrey, 2021; Hesselbarth et al., 2019; Hijmans, 2021). We separated landcover types identified on the annual crop inventory into natural, developed, rewarding, and non-rewarding agriculture since each category has shown different effects on insects in the past (Killewald et al., 2024). Rewarding agriculture included crops with flowers frequently visited by insects for floral rewards such as canola, flax, soybeans, peas, beans, and other vegetables. Non-rewarding agriculture included crops such as cereals. The natural category included land use types such as shrubland, forests, and wetlands. The developed category included anthropogenic structures such as buildings, roads, and towns. We extracted landscape composition as the percent of natural, developed, rewarding agriculture, and non-rewarding agriculture at six spatial scales (0.5, 1, 1.5, 2, 2.5, and 3 km) from the trap nests using the *lsm_c_ca* function from the *landscapemetrics* package (Hesselbarth et al. 2019). We tested the effects of landscape variables at different scales since insects often respond to different spatial scales (Karp et al. 2018). We extracted landscape configuration data as the edge density and landscape Shannon diversity using the *lsm_l_ed* and the *lsm_l_shdi* functions from the *landscapemetrics* package in R (Hesselbarth et al. 2019) at the six spatial scale described above.

5.3.4 Pollen analysis

Pollen removed from the trap nests was placed into a 1.5 ml centrifuge tube with 70% ethanol solution and processed following previous methods (Killewald et al., 2019; Westrich & Schmidt, 1986). Pollen was placed into a -20°C freezer after removal from trap nests. Slides were prepared by vortexing the pollen sample, pipetting a small amount onto a microscope slide, and mixing with sterile water to spread pollen grains on the slide. Amounts of the pollen sample were varied to keep concentration of pollen on the slides consistent. Slides were stained with fuchsin gel and a glass cover slip was placed on top.

Volumes of each pollen grains type were visually estimated (Folk 1951) using light microscopy under 400× magnification. Pollen was estimated to the lowest reasonable taxonomic level using Sawyer (Sawyer 1981). We also used a reference collection of over 250 species from 49 botanical families collected from Manitoba and other online resources such as the Isaacs lab reference collection (“Isaacs Lab at MSU,” 2017) and other identified images on the global pollen project website (Martin & Harvey, 2017) or PalDat (“PalDat- Palynological Database,” 2023). For particularly difficult to separate groups, such as Asteraceae and some Fabaceae, “type groups” were used (Sawyer 1981). Pollen type groups include multiple related taxa that are indistinguishable under light microscopy. The type groups described in this study largely follow Sawyer (1981).

We compared the pollen community collected by *Megachile* to the pollen community of blooming forbs identified from the local vegetation surveys (see Graham *et al.* 2020). Pollen is rarely identifiable to species using light microscopy. So the plant

species data from the vegetation surveys were used to create a matrix of pollen species by site matrix. To do this, we identified pollen samples from each blooming forb identified in the surveys to the lowest taxonomic level possible for light microscopy analysis. For example, *Aster* pollen is impossible to identify to species using light microscopy, so any *Aster* species identified from the floral survey was lumped into a group “Aster type” for this analysis.

5.3.5 Data analysis

Distance-based redundancy analysis (dbRDA) followed by permutational multivariate analysis of variance (PERMANOVA) were used to determine if the pollen communities collected between treatment groups were statistically different from each other. We also used ordinations to determine if the pollen community collected by *Megachile* was similar to the vegetation community present at the sites. Ordinations were made using the *capscale* function in the *vegan* package (Oksanen et al., 2019) and the Bray-Curtis dissimilarity measure. Ordinations were then tested with a PERMANOVA using the *adonis2* function using FDR corrected *p* values in R (R Core Team 2023; Oksanen et al. 2019). We set the number of permutations to 999 using the *permutations* argument in the *adonis2* function. Seeds were set before permutational analysis for reproducibility. If results from the PERMANOVA were significant, then a post hoc test was performed using the *pairwise.adonis* function (Martinez Arbizu 2017) to identify differences between treatment groups. Since significant results from PERMANOVAs can be due to differences in group centroids or group dispersions (Anderson 2017), we tested group dispersions separately with the *betadisper* and *permutest* functions from the *vegan*

package and using the TukeyHSD function from the stats package in R (Oksanen et al. 2019; R Core Team 2023). Unless stated otherwise in the results, dispersions of treatment groups were non-significant. We used the *multipatt* function from the indicpecies package in R (De Cáceres et al., 2023) to test if any pollen species were significantly associated with treatment types. The variables driving the ordinations were tested using the *envfit* function from the vegan package (Oksanen et al. 2019) and fitted to the plots based on significant effects ($p < 0.05$).

Generalized linear mixed-effect models (GLMM) from the glmmTMB package were used to test for differences in the pollen composition and nesting rates between treatment groups (Brooks et al., 2017). We tested if the pollen Shannon diversity, pollen richness, and the percent of Asteraceae, Brassicaceae, or Fabaceae were different between treatments or if these measures changed through time as a female constructed the nest. We did not test for differences between treatments for botanical families that made up <5% of total collection (Lythraceae) since sample size was low ($n=3$). Site was used as a random effect for all nest abundance models. Trap nest number was nested within site as a random effect for all pollen models. For percentage data, we examined gamma, beta, and Gaussian distributions and those that best fit the model based on Akaike Information Criterion score and log-likelihood were selected. We also examined Poisson and negative binomial (with linear or quadratic parameterization) distributions where count data were included in models. Since the beta distribution must have values between zero and one, but not exactly these values, we added or subtracted 0.001% from our pollen composition data before running the analysis. Similarly, the gamma distribution doesn't allow zeros, so we added 0.001% to our pollen composition data

before running the models. Hill-Shannon diversity indices for each sample were calculated using the *hill_taxa* function from the *hillR* package (Daijiang 2018). P-values for treatment effects were calculated using the *Anova* function from the *car* package (Fox & Weisberg, 2019). Tukey's honest significant difference test in the *glht* function was used to compare differences between treatment groups (Hothorn et al., 2008).

5.4 RESULTS

5.4.1 Nesting density

We collected 122 *Megachile* nests from all sites over the three sampling years (strip; $n=69$, control; $n=3$, natural; $n=50$). We collected significantly more nests per site/year in both strip (1.82 ± 0.53 nests / site-year, mean $\pm$ SE, GLMM: $df=1$, $z=2.90$, $p<0.001$, Figure 5.3) and natural (GLMM, 1.47 ± 0.58 , $df=1$, $z=3.31$, $p=0.003$) than control sites (0.1 ± 0.06). The strip and the natural sites had similar collection of nests (GLMM, $df=1$, $z=0.38$, $p=0.92$). The number of nests collected did not significantly change between years (GLMM, $df=2$ $X^2=1.21$, $p=0.55$). Overall, the number of cells per nest ranged from 1–16 (mean 7.34 ± 0.36) and females created more cells within nests at strip sites when compared to natural sites (GLMM, $df=1$, $z=2.84$, $p=0.012$). The number of cells per nest was similar between strip and control sites (GLMM, $df=1$, $z=1.05$, $p=0.54$). More nests were completed in semi-natural sites and floral strips than in conventional field margins.

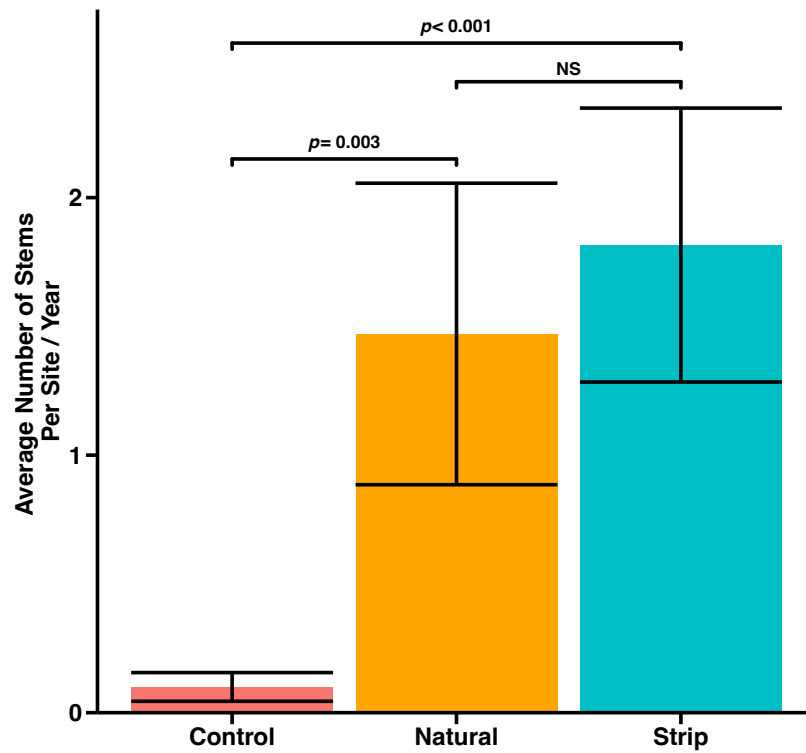


Figure 5.3 Bars graph shows average number of *Megachile* nests per site/year across the three treatment groups. Error bars represent $\pm$ one standard error. Nests were collected in Manitoba, Canada in the summer of 2019–2021. Treatments included strip; locations adjacent to a crop field with a floral strip, control; crop fields without a floral strip, natural; semi-natural sites that have not been managed for at least 5 years. Sample sizes: strip, $n=38$; control, $n=30$; natural, $n=34$.

5.4.2 Pollen resource use

Of the 122 collected *Megachile* nests, 39 nests contained pollen that had not been previously consumed by larvae (strip; $n=17$, control; $n=2$, natural; $n=20$). We removed a total of 93 pollen provisions (strip, $n=41$; control, $n=10$; natural, $n=42$). On average, we identified 2.38 ± 0.25 pollen samples per nest (range 1–6). Overall, pollen within the samples was identified to genus (75.43%) and 23.1% of pollen samples were Asteraceae or *Trifolium repens*-type, which were identified to type-groups based on their

morphology. We identified 13 pollen types across four botanical families (Table 5.1). The most common pollen identified overall were *Brassica* (37.8%), *Melilotus* (23.8%), and *Medicago* (8.1%). Strip sites contained every pollen identified in the study, except for *Rudbeckia* type, which was only identified in samples from the natural sites. Strip sites contained mostly *Melilotus* (31%) and *Brassica* (27%), which were not included in the sown seed mix, and *Medicago* (17%), which was included in the mix. Natural sites also had high levels of *Brassica* (56.4%) and *Melilotus* (12.9%), but also had substantial *Cirsium*-type pollen (15.4%). Natural sites contained nine of 13 pollen types identified from the study, where control sites only contained two—*Aster* type (60%) and *Melilotus* (39.5%). The amount of *Megachile*-collected pollen from plants identified in the vegetation surveys adjacent to the trap nests was not significantly different between treatments (GLMM, $df=2$, $X^2=4.47$, $p=0.11$). On average, 23.5% (± 4.28) of *Megachile* pollen loads were from plant species identified from the vegetation surveys.

Table 5.1 Percent composition of each pollen type identified from *Megachile* nests collected from three treatments. Nests were collected in Manitoba, Canada in the summer of 2019–2021. Treatments included strip; locations adjacent to a crop field with a floral strip, control; crop fields without a floral strip, natural; semi-natural sites that have not been managed for at least 5 years. Sample sizes: Control, $n= 10$; Natural, $n= 42$; Strip, $n= 41$.

Family	Pollen	Control	Natural	Strip
Asteraceae	<i>Aster</i> type	60	0	1.22
	<i>Centaurea</i> type	0	0.95	1.71
	<i>Cichorium</i>	0	2.74	9.76
	<i>Cirsium</i> type	0	15.43	0.85
	<i>Rudbeckia</i> type	0	1.55	0
	<i>Taraxacum</i> type	0	5.52	3.54
Brassicaceae	<i>Brassica</i>	0	56.43	27.85
Fabaceae	<i>Lotus</i>	0	0	0.49
	<i>Medicago</i>	0.5	0.48	17.76
	<i>Melilotus</i>	39.5	12.98	31.10
	<i>Trifolium pratense</i>	0	0	0.12
	<i>Trifolium repens</i> type	0	3.93	2.44
Lythraceae	<i>Lythrum</i>	0	0	3.17

The pollen community was significantly different between treatment groups (PERMANOVA, $df= 2$, $F=8.78$, $p<0.001$, Figure 5.4) with each treatment having a distinct community. An indicator species analysis returned no significant associations between pollen types and individual treatments. The pollen community inside nests was also distinct from the plant community observed at the sites (PERMANOVA, $F=11.843$,

$p < 0.001$, Figure 5.5). Shannon species diversity of pollen samples was not significantly different between treatments (GLMM, $df=2$, $X^2=3.83$, $p=0.15$) but was significantly different across years. Both pollen diversity (GLMM, $df=1$, $X^2=5.96$, $p=0.01$) and vegetation diversity (GLMM, $df=1$, $X^2=18.58$, $p < 0.001$) decreased as floral strips established. Although, vegetation richness significantly increased as floral strips established (GLMM, $df=1$, $X^2=20.23$, $p < 0.001$). Pollen Shannon diversity was not significantly associated with vegetation Shannon diversity (GLMM, $df=1$, $X^2=1.97$, $p=0.16$, Pearson $r=0.28$), but pollen diversity was inversely associated with vegetation richness (GLMM, $df=1$, $X^2=5.99$, $p=0.01$, Pearson $r=0.39$). Collection of the three most common pollen families, Asteraceae (GLMM, $df=1$, $X^2=0.47$, $p=0.79$), Brassicaceae (GLMM, $df=1$, $X^2=2.1$, $p=0.35$), or Fabaceae (GLMM, $df=1$, $X^2=1.79$, $p=0.41$), did not differ between treatments. Similarly, the pollen Shannon diversity (GLMM, $df=1$, $X^2=1.0$, $p=0.32$) and percent of Asteraceae (GLMM, $df=1$, $X^2=2.22$, $p=0.14$), Brassicaceae (GLMM, $df=1$, $X^2=0.10$, $p=0.75$), or Fabaceae (GLMM, $df=1$, $X^2=0.09$, $p=0.76$) in individual cells did not significantly change through time as the female constructed the nest.

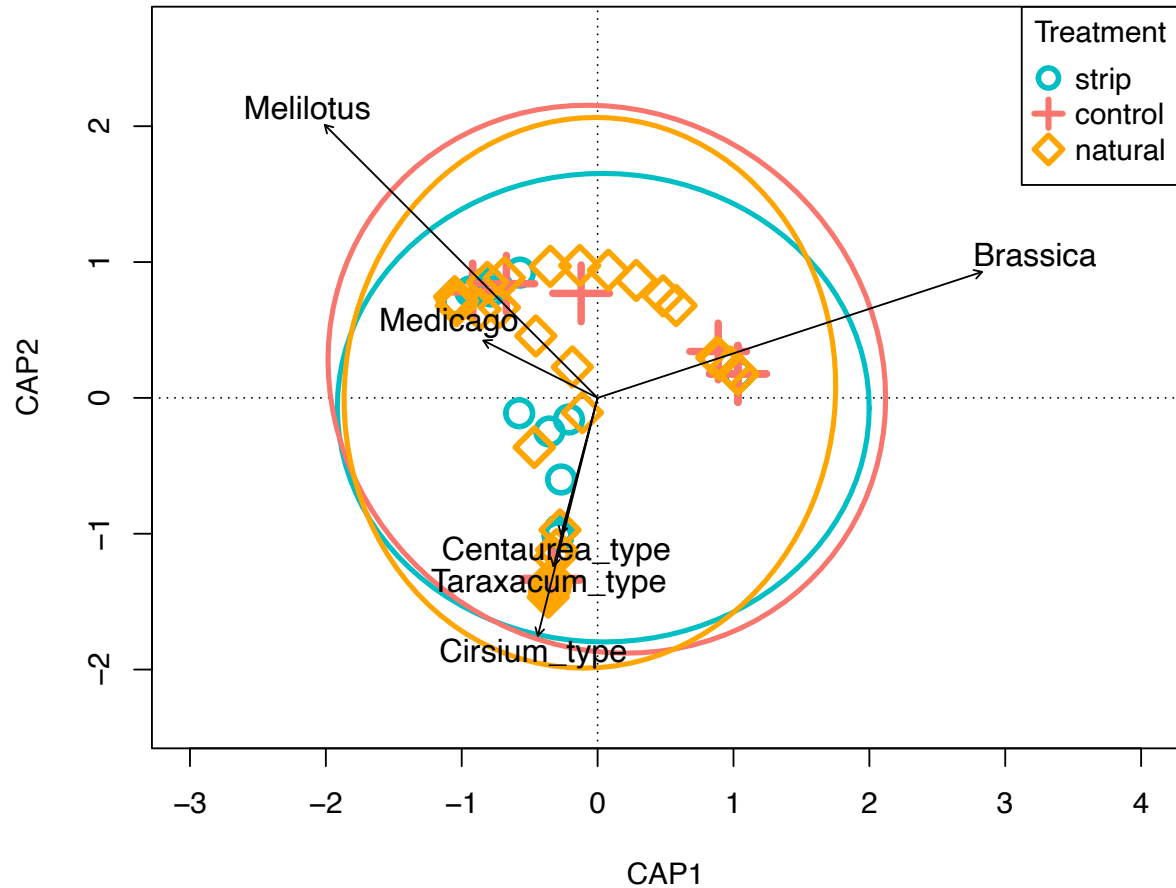


Figure 5.4 Ordination comparing community of pollen collected from *Megachile* nests from sites of three treatment groups. Ellipsis represents 95% confidence intervals around treatment groups. Vectors represent the significant ($p=0.05$) pollen types driving the ordination. Vector length represents the strength of the correlation between the species and the ordination. Some significant pollen types are not represented on the plot to increase readability. These pollen types have the same vector length and direction as *Centaurea* type pollen and include: *Chichorium* and *Aster* type. Treatments included strip; locations adjacent to a crop field with a floral strip, control; crop fields without a floral strip, natural; semi-natural sites that have not been managed for at least 5 years. Sample sizes: strip, $n=41$; control, $n=10$; natural, $n=42$.

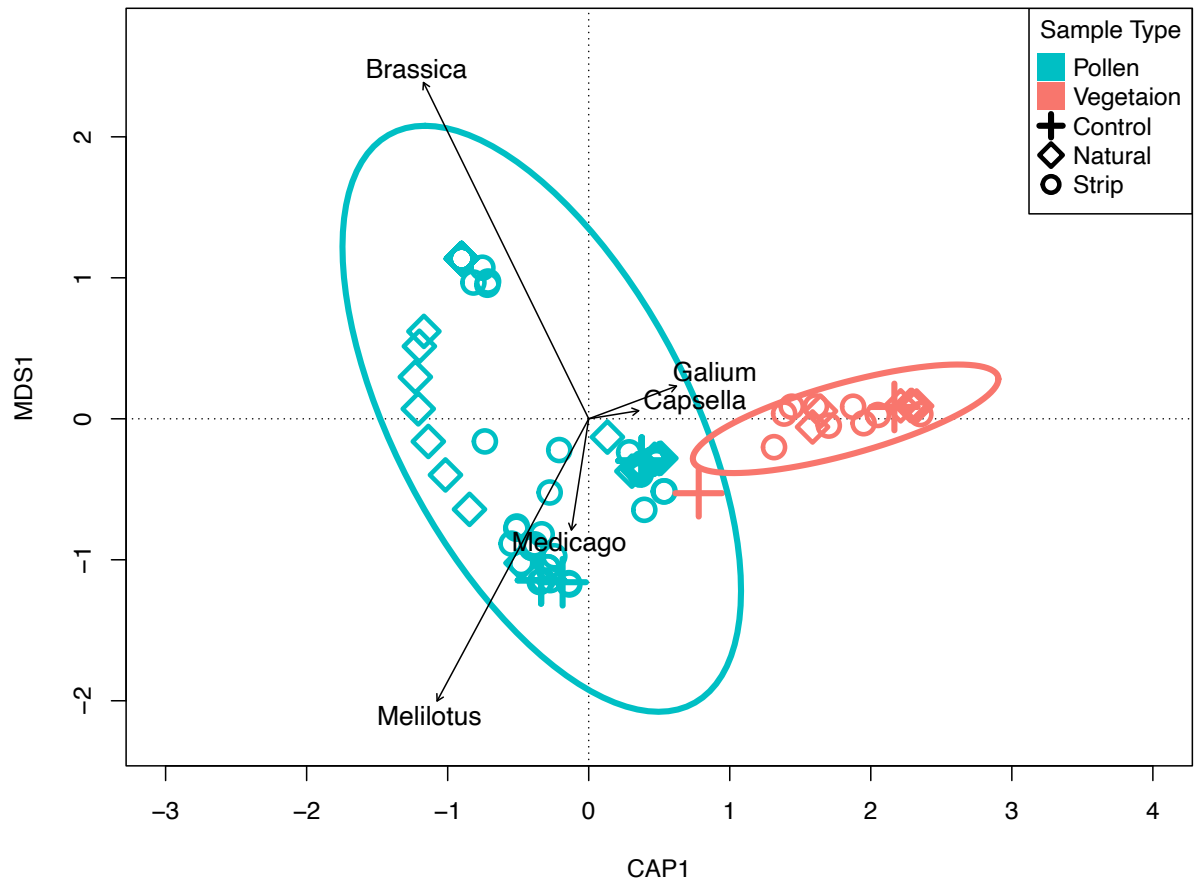


Figure 5.5 The pollen community from *Megachile* nests differs from the community found in vegetation surveys. The ordination compares community of pollen collected from *Megachile* nests and the pollen community of blooming plants surveyed from sites of three treatment groups. Ellipses represent 95% confidence intervals around sample groups. Vectors represent the significant ($p=0.05$) pollen types driving the ordination. Vector length represents the strength of the correlation between the species and the ordination. Some significant pollen types are not represented on the plot to increase readability. These pollen types have the same vector length and direction as *Galium* and include: *Anemonastrum*, *Onobrychis*, *Phacelia*, and *Fagopyrum*. *Helianthus* type pollen was also significant and had the same vector direction as *Galium*, but vector length was slightly shorter (CAP1 = 0.47). Treatments included strip: locations adjacent to a crop field with a floral strip; control: crop fields without a floral strip; and natural; semi-natural sites that have not been managed for at least 5 years. Sample types included pollen; pollen types identified from *Megachile* nests and vegetation; pollen types identified from the blooming plant community at the same sites as the *Megachile* pollen samples. Sample sizes: pollen strip, $n=41$; pollen control, $n=10$; pollen natural, $n=42$; vegetation strip, $n=9$; vegetation control, $n=2$; vegetation natural, $n=6$.

5.4.3 Species pollen use

We identified both adult bees and pollen samples from thirteen nests. Within these, we found three species of *Megachile*, *M. pugnata* ($n=3$ nests, $n=4$ pollen samples), *M. relativa* ($n=3$ nests, $n=7$ pollen samples), and *M. rotundata* ($n=7$ nests, $n=15$ pollen samples). We found that species collected different pollen communities (PERMANOVA, $df=2$, $F=2.73$, $p=0.027$, Table 5.2). However, we also found different dispersions of data between species, with *M. pugnata* having a different dispersion from both *M. rotundata* (Tukey, $p<0.001$) and *M. relativa* (Tukey, $p<0.001$). *Megachile pugnata* collected exclusively Asteraceae pollen while *M. relativa* collected pollens from Brassicaceae ($35\% \pm 20.88$) and Fabaceae ($65\% \pm 7.07$). *Megachile rotundata* collected the most pollen from Fabaceae ($38.2\% \pm 7.93$) and Brassicaceae ($33\% \pm 13.97$) but also collected Asteraceae ($20\% \pm 17.68$) and Lythraceae pollen ($8.67\% \pm 11.67$).

Table 5.2 Average ($\pm$ SE) percent composition of each pollen type identified from nests of three *Megachile* (Hymenoptera: Megachilidae) species. Nests were collected in Manitoba, Canada in the summer of 2019–2021.

Pollen type	<i>M. pugnata</i>	<i>M. relativa</i>	<i>M. rotundata</i>
<i>Aster</i> type	10 (0)	0	0
<i>Centaurea</i> type	15 (10)	0	0
<i>Cirsium</i> type	48.25 (29.67)	0	9 (27.5)
<i>Taraxacum</i> type	26.75 (16.42)	0	11 (27.54)
<i>Brassica</i>	0	35 (20.88)	33.13 (13.97)
<i>Medicago</i>	0	43.57 (7.48)	14.2 (7.39)
<i>Melilotus</i>	0	11.43 (10.14)	22.67 (16.31)
<i>Trifolium repens</i> type	0	10 (3.67)	0.67 (0)
<i>Lotus</i>	0	0	0.67 (0)
<i>Lythrum</i> type	0	0	8.67 (11.67)
sample size	4	7	15

5.4.4 Landscape

The density of nests collected per treatment was not significantly associated with any local scale variables. Nesting density significantly increased as the amount of rewarding agriculture at the largest, 3 km, spatial scale increased (GLMM, $df=1$, $X^2=6.09$, $p=0.014$). The number of cells per nest was inversely associated with Shannon vegetation diversity (GLMM, $df=1$, $X^2=10.71$, $p=0.001$), and both the amount of rewarding agriculture (GLMM, $df=1$, $X^2=25.48$, $p<0.001$) and edge density (GLMM, $df=1$, $X^2=21.77$, $p<0.001$) at the 3 km spatial scale. Pollen Shannon diversity was

positively associated with both forb cover (GLMM, $df=1$, $X^2=129.07$, $p< 0.001$) and blooming vegetation richness (GLMM, $df=1$, $X^2=9.56$, $p= 0.002$) but was not significantly associated with any landscape scale variables.

5.5 DISCUSSION

Floral strips have been shown to increase the abundance of pollinators (Albrecht *et al.*, 2021) but few studies have analyzed how these strips influence nesting rates and pollen collection of *Megachile* (Graham *et al.* 2020). To our knowledge, this is the first study to analyze how floral strips influence pollen selection and provisioning rates of *Megachile* in Canada (Sheffield *et al.* 2008). Nesting density of *Megachile* was significantly higher in strip sites when compared to control sites. In contrast, the number of cells per nest was similar between strip and control sites. Our results are supported by a previous study which also found significantly higher nesting of *Megachile* adjacent to floral strips (Graham *et al.* 2020). Graham *et al.* (2020) only found one of the 236 nests completed at a farm without a strip. Similarly, we found that most nests were completed at strip sites ($n=69$) while only three nests were collected at control sites across all sampling years.

We found that the pollen community collected by nesting *Megachile* was unique at all three treatments. Strip sites contained the most richness of pollen when compared to the other treatments. *Melilotus* was the second most commonly identified pollen overall in our study and was collected in high ($>10\%$) amounts in all three treatment groups. The collection of *Melilotus* was highest in the control sites (39.5%). The collection of

Brassica pollen was highest in the natural treatment and made up over 50% of overall pollen collections. Canola (*Brassica napus*, L.) is a widespread field crop of the Canadian prairies (Strelkov & Hwang, 2014) and volunteer mustards often inhabits roadsides and other areas in the landscape (M.K. Pers. Obs.). When collected, *Brassica* made up 70.7% (± 4.53) of the sample, although it may be a nutritionally limiting food source for bees (Filipiak 2019). It is possible that flowers in the semi-natural areas are limited, thus pushing bees to spill over into managed landscapes to meet nutritional demands (Montero-Castaño et al., 2016). Whenever *Brassica* pollen was found within a stem, there was also large amounts of canola within adjacent fields (> 30 ha within 1 km). The exception to this was one site in 2019, which had low amounts of canola within 1 km (1.5 ha), but this site also had low collection of *Brassica* pollen ($< 10\%$ of any sample). Although large scale flowering crops, such as canola, provide bees with temporary resources, they often have a net negative effect on pollinator abundance (Shaw et al. 2020). Bees, like many other organisms, need varied diets to effectively survive and reproduce (Goulson et al. 2015) and large-scale monocrop fields only provide floral resources for a short window during the growing season. Thus, season long floral resources are needed to effectively support pollinators (Williams et al., 2015).

Floral strips have proven to be effective at increasing resources for bees during the flight season and we hoped that *Megachile* would use the floral strips to provision for their offspring. Almost 30% of the pollen identified from strip sites was from plants included in the seed mix itself. *Medicago*, *Cichorium*, *Trifolium repens* type, and *T. pratense* (L.), were included in the seed mix planted in the strips (Killewald et al., 2024). The above pollen types, excluding *T. repens* type, were collected in higher abundances in

strip sites when compared to the other treatments. This suggests that *Megachile* are foraging within the strips to meet nutritional demands.

Strips not only established with sown species but included volunteer species as well. Some strips contained high amounts of *Melilotus* creeping in from the edges, which could be a source of the *Melilotus* pollen collected at the sites. When we compared the pollen community and the vegetation community present at individual sites, we found that *Megachile* pollen loads contained 23.5% of pollen types identified from the vegetation surveys. This means that on average, *Megachile* collected 23.5% of their pollen from identified plants directly adjacent to the nests, leaving over 75% of their pollen collection to other areas of the landscape. On average, the floral strips were 0.6 ha in size (Killewald et al., 2024) and a typical flight range of *Megachile* is 0.5–1.5 km (Greenleaf et al. 2007), meaning that the *Megachile* likely forage outside of the floral strips. However, some smaller *Megachile* may not fly far from their nests and seem to prefer to forage within 150 m, as floral resources located at this distance alter reproductive decisions (Peterson & Roitberg, 2006, 2016). It is not surprising that *Megachile* supplemented their diets with pollens available outside of the strips as a previous study by Graham et al. (Graham et al. 2020) found similar results. Although *Megachile* used pollen from nearby strip sites, a majority of pollen in their study came from invasive spotted knapweed (*Centaurea maculosa*, L.), which was commonly found near some of their sites (Graham et al. 2020). Spotted knapweed has been present in western Canada for decades (Lamarck & Asteraceae, 2001) and has recently spread to Manitoba (Invasive Species Council of Manitoba 2017). Although we have never personally seen this species at any of our sites during sampling, we did find low amounts

(< 2% overall, 10–40% of pollen sample when collected) of *Centaurea* type pollens from two sites. This may indicate that *Megachile* are well adapted to find preferred pollen resources in the landscape that are uncommon.

We had planned to identify adults from each individual nest, but unfortunately, dermestid damage made processing most of these samples impossible. Nonetheless, we managed to identify three bee species (*M. relativa*, *M. rotundata*, and *M. pugnata*) and associated pollen samples from thirteen nests. The three species we found are the first, third, and fifth most common stem or log-nesting leafcutter bees in Manitoba (Gibbs et al., 2023; Gibbs, 2024). *Megachile inermis* (Provancher) and *M. frigida* (Smith) are common species in the province that we did not identify from our traps, either because our sample size was too low, or perhaps because of habitat or nest substrate preferences, which made them less common in our sites. *Megachile pugnata* collected exclusively Asteraceae pollen, which is unsurprising given that it is a well-known Asteraceae specialist (Tepedino & Frohlich, 1982). *Megachile relativa* also had a somewhat narrow dietary breadth, collecting pollens from three types of Fabaceae and one type of *Brassica* (Table 1). *Megachile rotundata* collected the most types of pollen ($n=8$), but this species also had the most samples ($n=15$). Some of the less common pollen species (*Trifolium repens*-type and *Lotus*) were only present from one pollen sample of *M. rotundata*. Surprisingly, of the most common pollen collection from *M. rotundata* was *Brassica* (33%, Table 1) with 14% of pollen collections from *Medicago*. We expected to find more *Medicago* pollen within nests of *M. rotundata* given that this species is commonly used for alfalfa production (Pitts-Singer and Cane, 2011). Although alfalfa was part of our seed mix, canola is more abundant in the studied landscape than alfalfa (Agriculture and

Agri-Food Canada, 2021) so pollen selection here could be due to availability of resources.

Megachile responses to landscape variables were strongest at the largest, 3 km radius, spatial scale. Nesting density significantly increased as the amount of rewarding agriculture at 3 km increased, but the number of offspring per nest was negatively associated with this variable. The number of offspring per nest was also negatively associated vegetation diversity at the local scale. This likely means that *Megachile* create more nests with fewer offspring in landscapes with more blooming field crops and or a diverse local vegetation community. Though the causes for this are not clear, Tepedino and Frohlich (1982) found that nests with fewer cells had a higher percentage of female offspring. Thus, it is possible that in landscapes with additional floral resources, *Megachile* produce nests with fewer males and more females. If this is true, then increases in females could increase pollination potential of adjacent flowers as only female bees tend to collect pollen. We did not record the sex of individual cells and since our nest occupants were destroyed, we cannot assess this for our study. Further research is necessary to understand this trend. At the local scale, pollen diversity within nests was positively associated with forb cover and blooming forb richness. Unsurprisingly, *Megachile* collect a higher diversity of pollens when more floral species are present. It has been documented that declining species often have a limited floral diet (Kleijn & Raemakers, 2008; Wood et al., 2019). Given that higher pollen diversity can improve an individual's ability to survive external stresses such as diseases (Di Pasquale et al. 2013), we hypothesize that the increased floral diversity at strip sites may increase population health of *Megachile*, though this has not been tested directly here.

5.6 CONCLUSION

Although many studies have shown that floral strips can improve bee abundance and diversity (Holzschuh et al. 2012; Blaauw and Isaacs, 2014; Kordbach et al. 2020), few have shown how these strips can improve diet and nesting of *Megachile*, a genus with important native and managed pollinators (Graham et al. 2020). Our study shows that floral strips increase nesting rates and supplement pollen foraging of *Megachile* in Manitoba, as 30% of pollen from nests came from the strips. Landscape analysis has shown that *Megachile* create more nests with fewer offspring in areas with large-scale flowering crops and or high local vegetation diversity. Previous studies have found that nests with fewer cells tend to have more females (Tepedino & Frohlich, 1982), so it may be possible that increased vegetation diversity drives *Megachile* to lay more female offspring. Our results are supported by another study, conducted in a very different agricultural context, which also found significantly higher nesting rates of *Megachile* at strip sites (Graham et al. 2020). We hope that the increases in dietary diversity can improve the survival and reproduction of the next generation of *Megachile* and that these data can be used to inform conservation practices in our region.

5.7 ACKNOWLEDGEMENTS

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CHAPTER 6. GRAZING SUPPORTS MORE OLIGOLECTIC BEES THAN HAY FIELDS- FUNCTIONAL DIFFERENCES IN POLLINATOR COMMUNITIES AMONG PASTURE LANDSCAPES

This manuscript has been submitted to The Canadian Entomologist and is currently (as of 14 October 2025) under review. A version of this manuscript is included below and is formatted to fit thesis guidelines.

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6.1 ABSTRACT

1. Insects are in decline globally with results of meta-analysis linking these declines to intense agricultural production. Agriculture relies heavily on the ecosystem services provided by beneficial insects, so supporting their populations in these areas is of importance.

2. I tested for differences in bee communities between two common agricultural systems in Manitoba and an unmanaged (natural) control. Treatments included Graze: pastures that were grazed with cattle, Hay: hay fields that were planted with a pollinator enhanced seed mix, and natural: unmanaged areas in Manitoba, Canada.
3. I captured more bees at natural and grazed sites than hay sites. Grazed sites supported more oligolectic (plant specialist) bees, while natural sites collected more polylectic (plant generalist) bees. Bee abundance was positively associated with overall field size. Associations between bee functional traits (diet, nesting, and sociality) and environmental variables are discussed.
4. Our results indicate that grazing, rather than cutting fields for hay, may be an important management tool for conservation of oligolectic and ground-nesting bees. Polylectic and stem nesting bees may use natural sites more than agricultural landcover types. Thus, increasing the amount of natural landcover for insects should be a priority.
5. Hay fields here contained the least floral diversity and bee abundance compared to other treatments. The positive association with natural landcover coupled with the natural treatment collecting the highest abundances of bees in our study support the protection of natural lands for bee conservation.

6.2 INTRODUCTION

Many insects are in decline worldwide (Hallmann et al., 2017) with habitat loss considered to be one of the top contributing factors for many species (Wagner, 2020). In

the prairie region of North America, most land was converted into agricultural production during the time of Western expansion. In Manitoba, this began in the early 1800s, but conversion of land to agriculture rapidly increased in the 1870s when railways were constructed (Henderson and Koper, 2014). Today, less than 30% of historic prairie land exists in Manitoba and this amount continues to decrease in recent years (Alemu et al. 2020). The conversion of prairies has resulted in the decline of many insect species, with many listed as endangered in this area (Hall, Catling and Lafontaine, 2011). This is problematic because insects provide many ecological functions, such as pollination (Klein et al., 2007), biological control (Rutledge et al. 2004), and nutrient recycling (Belovsky and Slade, 2000). Bees are one of the most important pollinators because they directly collect pollen and nectar to feed their offspring, which often results in greater crop yield and quality (Garibaldi et al., 2013). Bee pollination services contribute to at least 25% of global food production (Klein et al., 2007). However, bee populations are in decline (Grixti et al. 2009).

Bees, like many organisms, require access to food, mates, and shelter to reproduce. Oligolectic bees collect pollen and or nectar from a narrow range of hosts. Their specialization makes them important for crop pollination– but their narrow dietary range can sometimes increase rates of population declines (Bartomeus et al., 2013; Wood et al., 2019) and increase vulnerability to global climate change (Böttcher 2023).

Bees may build nests in a variety of locations, including ground nesting (Antoine and Forrest, 2021), stem nesting (Krombein, 1967), and hive nesting species (Kwafo et al. 2023). Many of the ground and stem nesting species are solitary, meaning that they generally live their life alone, but many hive nesting species are also social- meaning that

they will build a colony with hundreds or even thousands of worker bees. These life history traits may make species more or less vulnerable to environmental changes (Caitano et al. 2020) but overall effects vary between locations (Bartomeus et al., 2018). Various contributing factors for bee population declines have been identified (Goulson et al., 2015) and causes of these declines are often numerous and compounded, resulting in a “death by a thousand cuts” as described by Wagner et al. (2021). Given the importance of bees for crop production, their increased rates of decline in agricultural areas, and the inconsistent landscape effects between regions, understanding ways to conserve their populations should be a priority.

Many of the native prairies have been lost (Samson and Knopf, 1996) and areas that are used for grazing may act as a managed substitute for a natural prairie. The agricultural region of Manitoba was historically prairies, but agricultural conversion has now used these areas for pastures. Grazed pastures may increase local vegetation diversity and insect abundance (Vulliamy, Potts and Willmer, 2006) and may act as a reserve of insects in areas with little natural vegetation (Morandin et al., 2007). Most bee conservation programs aim to change the local landscape for the benefit of pollinators. This can generally be accomplished by increasing floral abundance or nesting areas. Sowing seed mixes for pollinator conservation (Blaauw and Isaacs, 2014) or altering field management practices, such as grazing (Kimoto et al. 2012), may increase floral abundance of local species. The former generally results in increased pollinator abundances and diversities (Albrecht et al., 2021; Blaauw and Isaacs, 2014; Killewald et al., 2024; Kordbach et al., 2020), but the responses of bees to grazing practices are often inconsistent. Some studies find positive effects of livestock grazing on bee

abundance (Kruess and Tschardt, 2002; Vulliamy, Potts and Willmer, 2006; Hatfield and LeBuhn, 2007; Sjödin, 2007) while other studies find negative effects (Yoshihara et al., 2008; Harmon-Threatt and Chin, 2016) or effects that vary based on insect group (Kimoto et al. 2012). Grazing intensity also plays a role in vegetation structure, particularly if forbs are composed of unpalatable species (Milton and Dean, 1990; Mayer, 2004) as livestock grazing will allow less palatable species to persist. Given that insect responses to grazing practices are varied, analyzing bee population responses to grazing practices is important to understand how to best conserve insect communities in Manitoba.

In Manitoba, agriculture accounts for almost 50% of available land south of the 51st parallel- with roughly 25% of farms utilizing grazing practices (Agriculture and Agri-Food Canada, 2021). Grazed fields often divided into paddocks that are rotationally grazed during the growing season. If fields are not grazed then they are typically used for hay production, with alfalfa being the dominant species found in these areas (M.K. pers. obs). Many bees pollinate alfalfa (Stephen, 1955), but monoculture crops often result in lower abundance and diversity of pollinators (Kremen et al.2002). Increasing floral diversity in hay fields may support greater pollinator diversity. Designing and sowing a pollinator enhanced seed mix with livestock friendly forbs that produce resources for pollinators could be a win-win for producers and bees. This has not been thoroughly tested (Buri et al. 2014).

Land that is used for grazing and hay production, like many fields in Manitoba, are often expansive, with an average crop area per farm of > 1,000 acres (Agriculture and Agri-Food Canada, 2021). Bees are attracted to flowers and floral distribution is often

patchy across the landscape (Maron and Crone, 2006), especially if plant density is low (Fagan et al. 2005). Additionally, since bees are central place foragers (Sommer et al. 2022) and bee nests are often aggregated around suitable microclimates (Antoine and Forrest, 2021), accurately estimating bee density per unit area is often difficult.

Studies often sample bees along transects (Vulliamy et al. 2006; Holzschuh et al. 2012; O'Connor et al., 2019; Olynyk et al. 2021), but most studies do not report if transects are randomly placed (Holzschuh et al. 2012; Olynyk et al. 2021) or selectively placed for ease of access or floral diversity (O'Connor et al. 2019). Non-random transects may give a biased estimate of pollinator abundance per unit area. If transects are always placed in areas with a high floral density or a nesting aggregation, then overall field level floral abundance and or the associated bee abundances will be inflated. A randomized design (McFrederick and LeBuhn, 2006) should more accurately determine pollinator abundance in a given area, so long as there are sufficient samples. Several studies have compared the efficiency of various collection techniques (Rhoades et al., 2017; Prendergast et al., 2020), but few studies have directly compared random and selectively placed methods to determine if differences are large enough to cause issues. Thus, determining ways to accurately determine pollinator abundances when sampling should be a priority to researchers.

I designed a study located in Western Manitoba to 1) determine differences in bee abundance and functional traits between grazed sites, enhanced hay fields, and semi-natural areas (used as a control), 2) determine relationships between field and landscape scale variables on bee abundance, and 3) determine if the effect of transect selection (randomly or selectively placed transects) alters bee capture rates. I expect that

selectively placed transects will increase bee captures, but random transects may better reflect effects of local land-use and landscape.

6.3 METHODS

6.3.1 Site selection

Thirteen sites were selected within the pasture agricultural area of western Manitoba (Figure 6.1). Of these sites, five were grazed by cattle (graze treatment), five were planted with a pollinator-friendly hay mix designed by the Xerces Society (hay treatment, Table S 7.7), and three were areas that were unmanaged for at least five years prior to sampling (natural treatment). Only one of these three treatments (hay sites) were physically manipulated (vegetation terminated and a new seed mix planted) to alter the vegetation at the site.

The hay treatment included sites where the existing vegetation was terminated and a pollinator enhanced seed mix was planted by the farmers at least a year prior to sampling (summer 2018). Traditional hay fields contain mostly alfalfa whereas the “pollinator enhanced seed mix” used here contained multiple species of legume and grass forages (Table S 7.7). Methods for sowing the seed mix were selected by the farmers based on best practices for their field and soil conditions. Established stands were dominated by alfalfa, which comprised 63% of the forage seed mix (Table S7.7). Farmers were instructed to only cut hay treatment fields after the alfalfa in the stand had finished blooming and to leave a strip of uncut vegetation to provide a refuge for pollinators . Farmers generally cut hay fields twice per season and sampling was conducted prior to

cutting. The hay treatment was the only treatment in the study where plants in the field were cut, bailed, and removed from the field.

In the graze treatment, cattle were typically rotationally grazed during the summer growing season (generally May–September) with one week on and three weeks off of pastures. Most cattle producers had roughly 30–50 cow-calf pairs per field. Sampling took place when the paddocks were not actively being grazed. The grazed treatment was the only treatment in the study that included livestock.

Natural areas were not managed (not grazed or cut for hay). Most of the natural areas were wildlife management areas which are open to the public (Malaher and Broomhill wildlife management areas), with the exception of one natural area- which was a privately owned 32-hectare pasture that had not been grazed or managed for the last decade. Vegetation at all three natural area treatments were primarily grasses and forbs with some woody species.

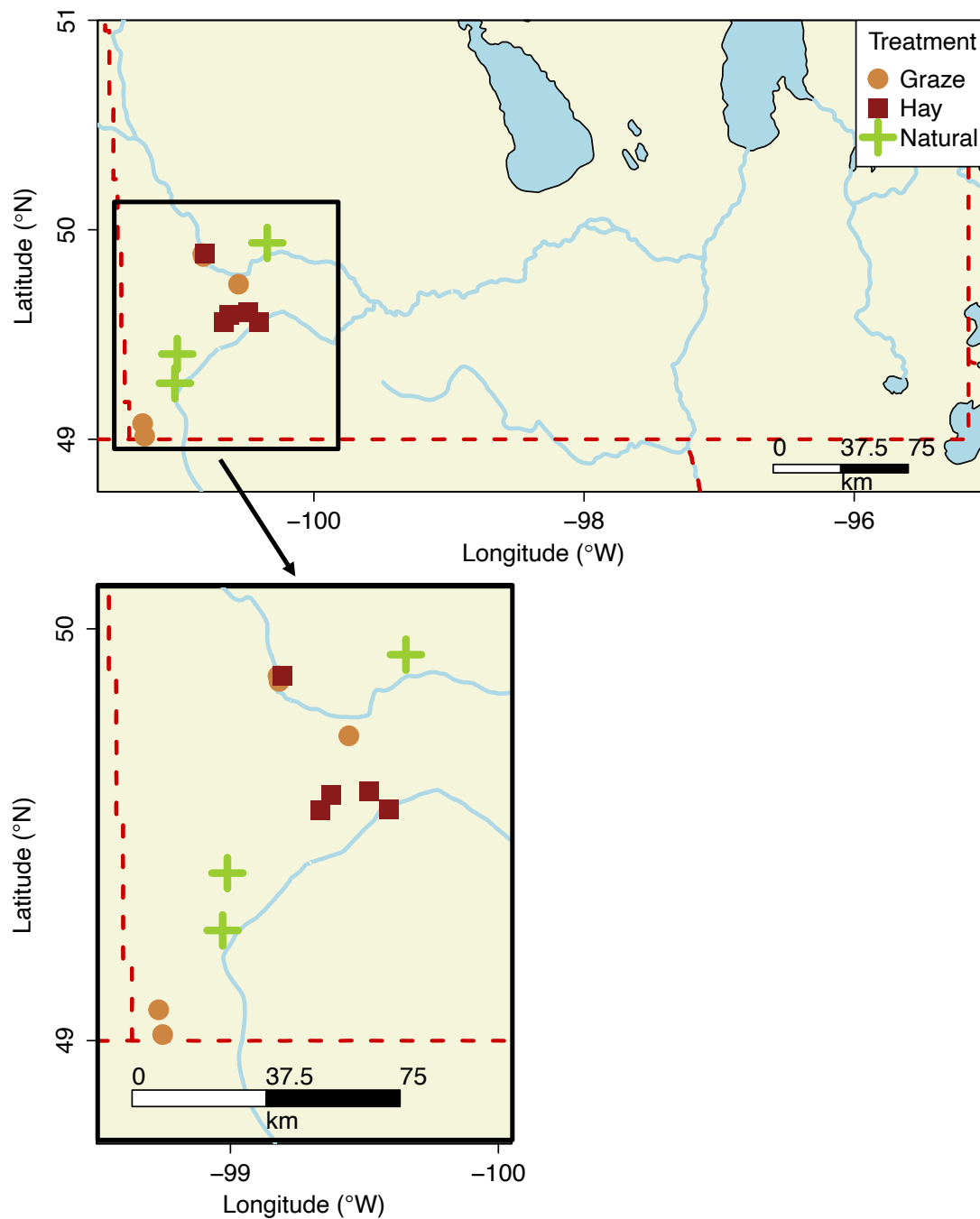


Figure 6.1 Locations of sites that were sampled in Manitoba, Canada in the summers of 2020 – 2022 for bees. Treatments included graze; locations that are rotationally grazed by cattle, natural; semi-natural sites that have not been managed for at least 5 years, and hay; fields that had been planted with an enhanced hay mixture. Dashed red lines represent borders between Manitoba, Ontario, Saskatchewan, North Dakota, and Minnesota. This figure was created using the PBSmapping package in R (Schnute, Boers and Haigh, 2021; R Core Team, 2023). Sample sizes: graze, $n=5$; natural, $n=3$; hay, $n=5$.

6.3.2 Targeted pollinator netting

Bees were collected along six, 50 m transects for 10 minutes per transect. Timers were paused while insects were transferred to vials so search time was standardized between transects. Insects were separated based on transect type (random or targeted), transect number (1–3), and host plant. Two different types of transects were used in this study: random and targeted. In order to estimate pollinator abundance at the field level, randomly selected transects (random treatment) were used (McFrederick and LeBuhn, 2006). Prior to visiting the fields, two hundred random GPS coordinates were selected and a new sampling location was selected at every visit. Coordinates located in water or in the middle of a herd of cattle were not sampled and instead, the next coordinate on the list was selected. A random number generator was used to select a direction for each transect (0–360) and a compass was used to verify transect direction prior to sampling. At each field we also used targeted transects (targeted treatment) where we would place transects within high floral abundance areas, preferably where bees were visible. Targeted transects were used to capture the highest diversity of pollinators possible per site. Three targeted and three random transects were used to capture bees at each site. Net transects were conducted between 12 July and August 10 from 2020–2022. I collected an average of $1.15 (\pm 0.06)$ samples per site/year and processed a total of 234 transect samples. Sampling generally occurred between management activities at hay and grazed sites. Sampling at grazed sites took place while cattle were not actively grazing the sampling areas. Sampling at hay sites occurred between cuttings of hay.

6.3.3 Specimen processing

Bees were pinned and identified to species. Taxon concepts were based on taxonomic literature and were referenced against expertly identified specimens. Bees were first identified to genus (Packer, Genaro and Sheffield, 2007) and then to species using described taxon concepts: Andrenidae (LaBerge, 1967, 1969, 1971, 1973, 1977, 1980, 1986, 1987, 1989; Ribble, 1967, 1968, 1974; Shinn, 1967; LaBerge and Bouseman, 1970; LaBerge and Ribble, 1972, 1975; Bouseman and LaBerge, 1979; LaBerge and Thorp, 2005), Apidae (LaBerge, 1956b, 1956a, 1961; Lavery and Harder, 1988; Droege et al., 2010; Rehan and Sheffield, 2011; Williams et al., 2014; Droege and Rehan, 2016; Onuferko, 2017, 2018), Colletidae (Stephen, 1954; Snelling, 1966, 1968, 1970, 1983), Halictidae (Sandhouse, 1941; McGinley, 1986, 2003; Coelho, 2004; Gibbs, 2010, 2011; Gibbs et al., 2013), and Megachilidae (Sandhouse, 1939; Michener, 1947; Baker, 1975; Sheffield et al., 2011; Gonzalez and Griswold, 2013; Semmler et al., 2017).

6.3.4 Vegetation and weather data

Vegetation and weather data were collected on the same days as the targeted pollinator netting and were used as variables in analyses. Floral surveys were conducted at each transect described in the targeted pollinator netting section above. I placed a 1 x 0.5 m quadrat every 5 m along the transect (n=11 quadrats per transect) and used published figures (Folk 1951) to visually estimate the percent ground cover of forbs, grasses, and bare ground per quadrat. I also visually estimated the cover of each blooming forb species per quadrat. Identifications of blooming forbs were completed with the use of published guides (Scoggan, 1957; Newcomb, 1989) and blooming forbs

were generally identified to species, but sometimes genus where species identity was uncertain.

At each sample, I also collected local weather data. Cloud cover (sunny, partly cloudy, or overcast) was visually estimated and average and maximum wind speeds and ambient air temperature was recorded with a Kestrel 3000 pocket weather meter (Philadelphia, PA). Weather data were used in base models described below and vegetation data were used in models to determine effects of local vegetation on bee community assemblages.

6.3.5 Landscape data

I extracted both landscape composition and landscape configuration data using R version 4.3.1 (R Core Team 2023). Landscape data were extracted from the appropriate annual crop inventory available online from Agriculture and Agri-Food Canada (Agriculture and Agri-Food Canada, 2021). I used packages in R such as raster, landscapemetrics, and sampSurf to extract and summarize landscape data from the above data layers (Hesselbarth et al., 2019; Gove, Jeffrey, 2021; Hijmans, 2021).

I separated landscape composition data into four categories that have shown varying effects on insect populations in Manitoba (Killewald et al., 2024). Categories included: melittophilous crops such as canola, sunflower, alfalfa, etc. ($n=34$ landcover types), anemophilous crops, such as wheat, oats, etc. ($n=25$), developed (urban centres, buildings, roads, etc., ($n=2$)), and natural landcover types (grasslands, forests, wetlands, etc., ($n=10$)). I extracted landscape composition data as the percent of natural, developed, melittophilous, and anemophilous crops at six spatial scales (0.5, 1, 1.5, 2, 2.5, and 3

kilometres (km)) from each site using the *lsm_c_ca* function from the *landscapemetrics* package (Hesselbarth et al. 2019). I extracted data at multiple spatial scales because insects show varying responses to landscape variables (Karp et al., 2018d). I extracted landscape configuration data as the landscape Shannon diversity and edge density of landcover types using the *lsm_l_shdi* and *lsm_l_ed* functions, respectively, (Hesselbarth et al. 2019) at the six spatial scale described above. Landcover types were summarized into major groups (melittophilous crops, anemophilous crops, urban, and natural) for the landscape configuration variables. Landscape composition and configuration data at the six spatial scales were then used in later models described below.

6.3.6 Data analysis

Generalized linear mixed-effect models (GLMM) from the *lme4* package were used to test for differences in bee abundances between treatment groups (D. Bates et al., 2015). I also compared the effects of field size in separate models. Site was used as a random effect for all models. I examined Poisson and negative binomial distributions using the *glmer* and *glmer.nb* functions respectively. P-values for treatment effects were calculated using the *Anova* function from the *car* package (Fox and Weisberg, 2019). The *emmeans* function from the *emmeans* package (Lenth 2022) was used to test for differences between pairs of treatment groups.

I tested for local (field scale) and landscape level effects on bee abundances by creating a base model that included sampling day of year, latitude and longitude decimal coordinates of sites, time of sample, and the weather variables described above. I retained significant variables and then used this base model to compare base plus local and base

plus landscape level variables. Base plus local models included local scale variables described above such as the transect-level percent of forb, grass, and bare ground cover as well as blooming forb variables. Base plus landscape models were first created by selecting the appropriate spatial scale for the data. I included all landscape level variables for each spatial scale into separate models and selected the spatial scale with the lowest AIC score. I then retained significant variables from this spatial scale for our final landscape model.

Distance based redundancy analysis (dbRDA) and permutational analysis of variance (PERMANOVA) were used to determine compositional differences between treatment groups. I also used ordinations to visually compare the bee communities between treatment groups. Ordinations were made using the *capscale* function in the *vegan* package (Oksanen et al. 2019) using the Bray-Curtis dissimilarity measure. Ordinations were then tested with a PERMANOVA using the *adonis2* function using FDR corrected *p* values in R (R Core Team 2023; Oksanen et al. 2019). I set the number of permutations to 999 using the *permutations* argument in the *adonis2* function. Seeds were set before permutational analysis for reproducibility. If results from the PERMANOVA were significant, then a post hoc test was performed using the *pairwise.adonis* function (Martinez Arbizu 2017) to identify differences between treatment groups. Since significant results from PERMANOVA can be due to differences in group centroids or group dispersions (Anderson 2017), I tested group dispersions separately with the *betadisper* and *permutest* functions from the *vegan* package and using the TukeyHSD function from the *stats* package in R (Oksanen et al. 2019; R Core Team 2023). Unless stated otherwise in the results, there were no significant differences in

dispersions between treatment groups. The variables driving the ordinations were tested using the *envfit* function from the *vegan* package (Oksanen et al. 2019) and fitted to the plots based on significant effects ($p < 0.05$).

We used the fourth corner analysis to determine relationships between species and environmental traits. We calculated the fourth corner statistic using the *ade4* package in R (Dray & Dufour, 2007). Local and landscape variables included those described above as well as treatment. We selected the landscape scale that bees responded to best in the insect abundance GLMMs outlined above (3 km). Bee functional traits included nesting, lecty, and sociality (Cane, 1987; Gibbs et al., 2023). We used the FDR correction for multiple comparisons and seeds were set prior to running the analysis for reproducibility.

6.4 RESULTS

6.4.1 Insect data

I collected a total of 564 individual bees from five families and 70 different species (Table S 7.8). Of these, 227 were collected along the random transects ($n=117$) and 337 were collected along the targeted transects ($n=117$). Apidae were most abundant ($n=292$) followed by Andrenidae ($n=159$), Megachilidae ($n=65$), Halictidae ($n=31$), and Colletidae ($n=17$). I collected 373 polylectic and 173 oligolectic bees. Fourteen of seventeen oligolectic species specialize on various members of the Asteraceae. Sixteen

individuals were parasitic species and thus, do not collect pollen for their offspring. Two cleptoparasitic species, *Triepeolus obliteratus* Graenicher and *T. occidentalis* (Cresson), likely attack Asteraceae specialists in the genus *Melissodes*. Eusocial ($n=274$) and solitary ($n=272$) bees were roughly equal in number.

6.4.2 Insect community

The species composition of sites was significantly different between treatments (PERMANOVA, $df=2$, $F=3.06$, $p<0.001$, Figure 6.2). The hay and natural treatments were distinct from each other (PERMANOVA, $df=1$, $F=5.74$, $p=0.003$) while all other comparisons were similar. These two treatments had significantly different dispersions (Tukey, $p=0.001$) and thus had different levels of variation within sites of each group.

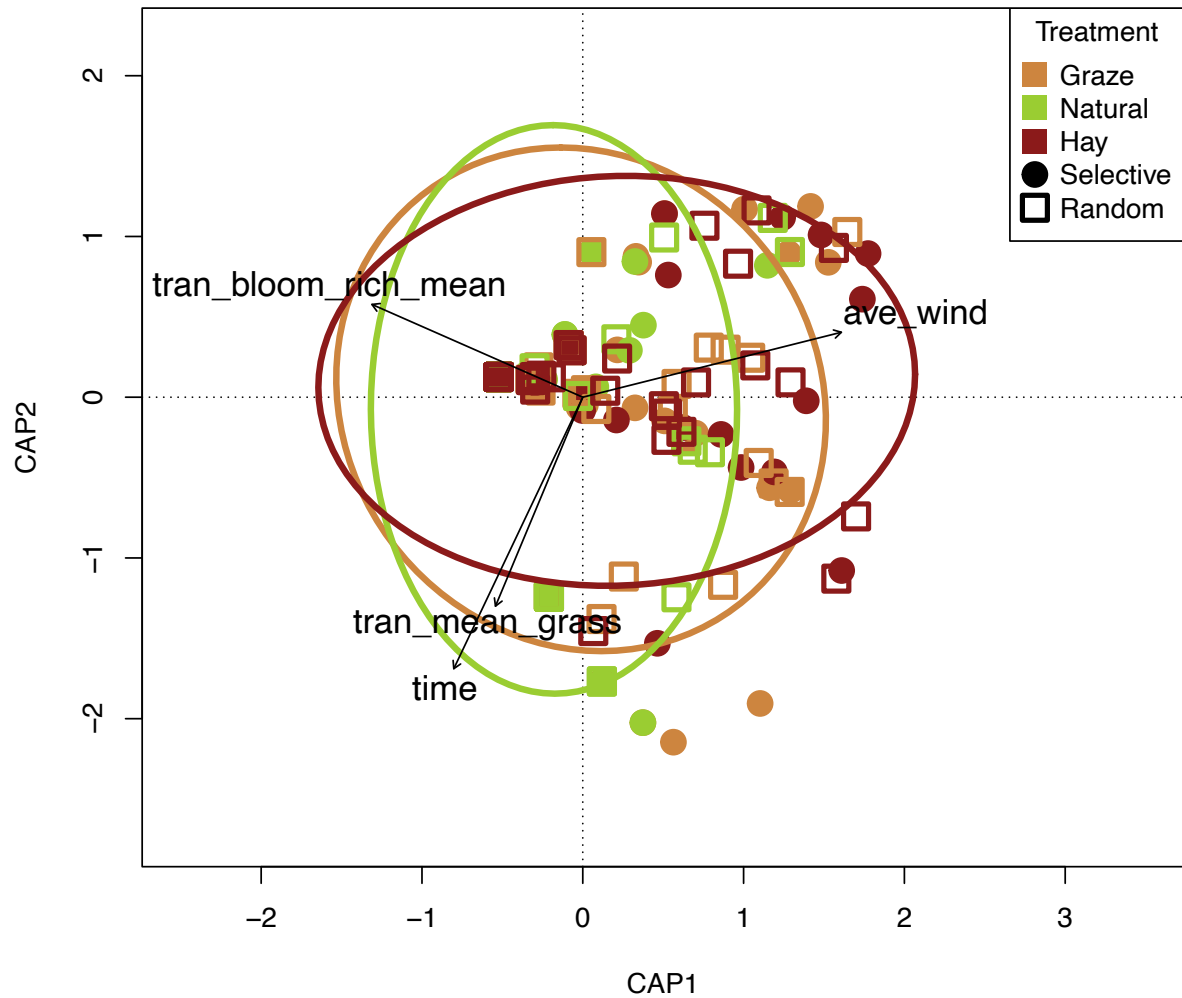


Figure 6.2 Ordination of a distance-based redundancy analysis with Bray-Curtis dissimilarity of bee species composition at sites from three treatments. Ellipses represent 95% confidence intervals around treatment groups. Vectors represent the significant ($p=0.05$) variables driving the ordination. Vector length represents the strength of the correlation between the species and the ordination. *Tran_bloom_rich_mean*; average blooming species richness per transect, *ave_wind*; average wind speed during the sampling time, *tran_mean_grass*; average transect level grass cover, *time*; time of day that sampling started. Treatments included graze; locations that are rotationally grazed by cattle, natural; semi-natural sites that have not been managed for at least 5 years, and hay; fields that have been planted with an enhanced hay mixture. Sample sizes: graze, $n=16$; natural, $n=12$; hay, $n=11$.

6.4.3 Bee capture rates

Targeted transects captured marginally more bees (2.88 ± 0.42 bees per transect) than random transects (1.94 ± 0.40) (GLMM, $df=1$, $X^2= 3.43$, $p= 0.06$). Bee capture rates were significantly different between treatments (Table 1, Figure 3D, Figure S 7.14). These results held true when comparing bees between treatments with random transects only, targeted transects only, and both transect types. Hay fields captured the least number of bees (0.61 ± 0.12) compared to both grazed (GLMM, 2.81 ± 0.54 , $df=1$, $z= 4.77$, $p< 0.0001$) and natural sites (GLMM, 3.53 ± 0.56 , $df=1$, $z= 5.22$, $p< 0.0001$). Capture rates of bees at grazed and natural sites were similar. Bee capture rates were also significantly different between years (Table 1) but no interaction existed between treatment and year. I collected significantly more bees in 2021 (3.47 ± 0.79) than 2020 (GLMM, 1.50 ± 0.32 , $df=1$, $z= 2.94$, $p= 0.009$) but other comparisons showed no difference.

6.4.4 Functional traits

I found associations between environmental traits, species traits, and bee capture rates using generalized linear mixed effect models and the fourth corner analysis. Specifically, I found both positive and negative correlations between landscape variables and treatment on bee traits such as lecty and nesting. Results for the fourth corner analysis are listed in appropriate subsections for each species trait below.

6.4.4.1 *Lecty*

Oligolectic bees were more common at grazed (GLMM, 1.39 ± 0.5 , $df=1$, $z= 3.9$, $p= 0.0009$) and natural (GLMM, 0.52 ± 0.2 , $df=1$, $z= 3.18$, $p= 0.018$, Figure 6.3A, Table 6.1) sites than hay sites (0.02 ± 0.02). Polylectic bees were most common at natural sites (2.7 ± 0.47) than hay sites (GLMM, 0.59 ± 0.11 , $df=1$, $z= 3.5$, $p= 0.0058$). Hay and natural fields captured significantly more polylectic (hay: 0.59 ± 0.11 , natural: 2.77 ± 0.47) than oligolectic bees (hay: GLMM, 0.015 ± 0.015 , $df=1$, $z= 3.49$, $p= 0.006$; natural: GLMM, 0.52 ± 0.2 , $df=1$, $z= 5.2$, $p< 0.0001$). I detected no differences between oligolectic and polylectic bees at grazed sites.

The fourth corner analysis found that oligolectic bees were positively correlated with the amount of natural land cover and negatively correlated with non-rewarding crops at 3 km. Additionally, polylectic bees were positively associated with non-rewarding crops at 3 km (Table 6.2).

6.4.4.2 *Nesting*

Stem nesters were more common in natural sites (0.72 ± 0.17) than grazed (GLMM, 0.18 ± 0.05 , $df=1$, $z= 3.16$, $p= 0.04$, Figure 6.3B, Table 6.1). Ground-nesting bees were more common at grazed (GLMM, 1.68 ± 0.51 , $df=1$, $z= 5.02$, $p< 0.0001$) and natural sites (GLMM, 0.84 ± 0.18 , $df=1$, $z= 4.11$, $p= 0.001$) than hay sites (0.03 ± 0.02). Hive nesting bees were equally as abundant among the three treatments. Grazed sites collected significantly less stem nesters than both ground (GLMM, $df=1$, $z= 5.82$, $p< 0.0001$) and hive nesting (GLMM, 0.92 ± 0.2 , $df=1$, $z= 4.51$, $p= 0.0002$) bees, but

abundances of hive and ground-nesting bees were similar. Hay sites collected more hive than ground nesters (GLMM, 0.03 ± 0.02 , $df=1$, $z= 3.7$, $p= 0.006$) while all other comparisons were similar. Natural sites collected equal numbers of bees from each nesting category.

Ground-nesting bees were negatively associated with both Shannon landscape diversity and non-rewarding agriculture at 3 km (Table 6.2). The fourth corner analysis found that hive nesting bees were positively associated with natural sites- but their capture rates were similar between treatments (Table 6.1).

6.4.4.3 Sociality

Social bees were more common at natural (1.87 ± 0.42) than hay sites (GLMM, 0.57 ± 0.11 , $df=1$, $z= 3.02$, $p= 0.03$, Figure 6.3C, Table 6.1) while other comparisons of social bees were similar. Solitary bees were more common at graze (GLMM, 1.79 ± 0.49 , $df=1$, $z= 5.07$, $p< 0.0001$) and natural (GLMM, 1.41 ± 0.29 , $df=1$, $z= 4.81$, $p< 0.0001$) than hay sites (0.03 ± 0.02). Grazed and natural sites captured equal numbers of eusocial and solitary bees while hay sites captured more social (0.57 ± 0.11) than solitary bees (GLMM, $df=1$, $z= 3.77$, $p= 0.002$).

The fourth corner analysis found strong positive associations between social bees and both natural and hay treatments (Figure 6.3).

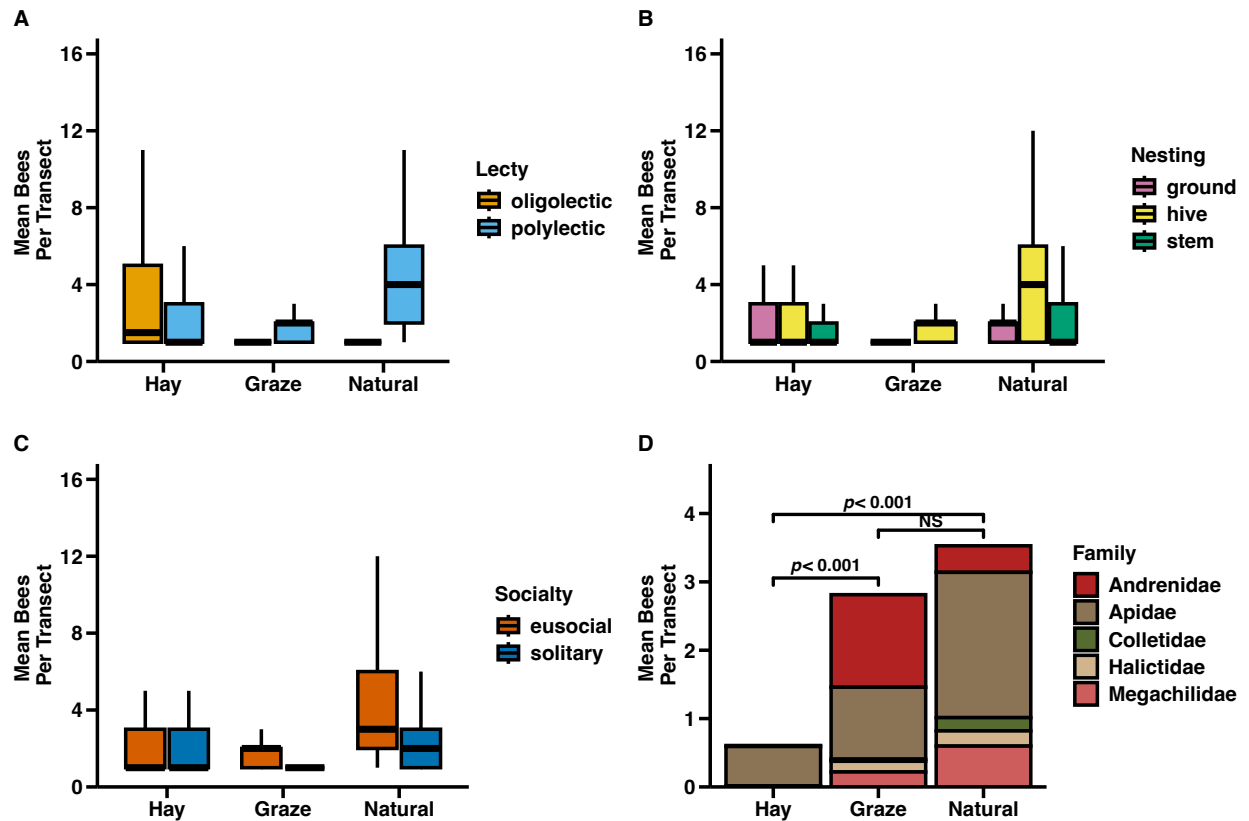


Figure 6.3 Boxplots (A, B, C) and stacked bar graph (D) showing the average number of bees collected per transect based on traits (A, B, C) and family composition (D) at sites of three treatments. *P* values indicate results from a generalized linear mixed effect model testing the differences in bee abundance between treatments with site as a random effect. Data were collected in Manitoba, Canada in the summers of 2020 – 2022 and plot was created with ggplot in R (Wickham, 2016; R Core Team, 2023). Treatments included hay; fields that have been planted with an enhanced hay mixture, graze; locations that are rotationally grazed by cattle, natural; and semi-natural sites that have not been managed for at least 5 years. Sample sizes: hay, $n=11$; graze, $n=16$; natural, $n=12$.

Table 6.1 Statistical results of generalized linear mixed effect models of bee capture rate data collected at sites of three treatments in Manitoba, Canada in the summers of 2020–2022. Treatments included hay: fields that were planted with an enhanced hay mixture and were cut for livestock feed; graze: fields that were rotationally grazed by cattle; and natural: locations without any management and were unmanaged for at least five years prior to sampling. Variables included Lecty, oligolectic and polylectic categories; Sociality, solitary and eusocial categories; and nesting, ground (included ground and surface), nest (included nest and cavity), and hive. Site was used as a random effect for all models.

		Treatment	Year	Lecty	Sociality	Nesting
Variable	df	2, 232	2, 232	1, 467	1, 467	2, 700
	X ²	41.36	11.42	25.57	0.78	29.37
	<i>p</i>	> 0.0001	0.003	> 0.0001	0.37	> 0.0001
Variable * Treatment	df		4, 230	2, 466	2, 466	4, 698
	X ²		6.07	14.98	15.66	28.92
	<i>p</i>		0.19	0.0005	0.0004	> 0.0001

Table 6.2 Statistical results of a fourth corner analysis of bee capture rate data, bee species traits, and environmental traits collected at sites of three treatments in Manitoba, Canada in the summers of 2020–2022. Treatments included hay, fields that were planted with an enhanced hay mixture and were cut for livestock feed; graze, fields that were rotationally grazed by cattle; and natural, locations without any management and were unmanaged for at least five years prior to sampling. Environmental variables included natural, percent of natural landcover within a 3 km radius of sites; nonrewrd.ag, percent of non-rewarding agriculture within a 3 km radius of sites; landscape diversity, landscape Shannon diversity calculated at the 3 km spatial scale surrounding sites; natural, natural treatment category; hay, hay treatment category. Species traits included lecty: oligolectic or polylectic; nesting: ground or hive; and sociality: social or solitary. Variable comparisons were included if the adjusted $p < 0.1$ as these may be ecologically relevant associations.

variables		observed values	standardized values	p	adjusted p
natural	oligolectic	0.37	2.519	.009	0.049
nonrewrd.ag	oligolectic	-0.535	-2.922	.001	0.032
nonrewrd.ag	polylectic	0.465	2.451	.001	0.032
nonrewrd.ag	ground	-0.451	-2.359	.002	0.043
landscape diversity	ground	-0.315	-2.161	.005	0.087
natural	hive	137	5.362	.001	0.032
natural	social	136	3.415	.001	0.032
hay	social	38	-2.308	.019	0.087

6.4.5 Local and landscape effects on bee abundances

Bees did not respond significantly to any local (field scale) vegetation variables but were positively associated with the percent of natural landcover at the largest, 3 km, spatial scale (GLMM, $df=1$, $X^2= 6.36$, $p= 0.012$). I found that the number of bees

collected per transect was positively associated with field size (GLMM, $+0.00544 \pm 0.00148$ bees / hectare, $df=1$, $X^2= 13.47$, $p < 0.001$). There was no interaction between field size and treatment or transect type.

I found that hay sites contained the most forb cover compared to both grazed (GLMM, $df=1$, $t=3.69$, $p= 0.007$) and natural sites (GLMM, $df=1$, $t=3.75$, $p= 0.01$, Table 6.3). Hay sites were dominated by alfalfa and contained the highest forb cover.

Unfortunately, hay sites also contained the least floral diversity compared to both grazed (GLMM, $df=1$, $t=5.3$, $p= 0.0008$) and natural sites (GLMM, $df=1$, $t=4.44$, $p= 0.003$). Hay sites were dominated by alfalfa as this single species made up 99.3 % ($\pm 0.91\%$) of observed forb cover (63% of sown seed mix) at hay sites. Multiple species of forbs were included in the seed mix sown at hay sites, but only alfalfa established. Both forb cover and floral Shannon diversity was similar between grazed and natural sites. Floral community structure was slightly different between sites. I found more blooming Asteraceae at grazed than hay sites (GLMM, $df=1$, $t=3.7$, $p= 0.006$) but hay sites contained more blooming Fabaceae than either grazed (GLMM, $df=1$, $t=7.1$, $p= 0.0001$) or natural sites (GLMM, $df=1$, $t=5.96$, $p= 0.0004$).

Table 6.3 Average (SE) values of Shannon diversity and percent cover of forbs, grasses, bare ground, blooming Asteraceae, and blooming Fabaceae at sites of three treatments in Manitoba, Canada in the summers of 2020–2022. Treatments included hay, fields that were planted with an enhanced hay mixture and were cut for livestock feed; graze, fields that were rotationally grazed by cattle; and natural, locations without any management and were unmanaged for at least five years prior to sampling. Data were collected along six vegetation belt-transects per site that were 50 m long and sampled every 5 m.

Treatment	Shannon diversity	forb	grass	bare	Asteraceae	Fabaceae
graze	2.61 (0.07)	42.19 (1.007)	52.97 (1.09)	4.83 (0.63)	1.77 (0.28)	0.34 (0.8)
hay	1.05 (0.014)	65.04 (2.0)	25.71 (1.54)	9.25 (0.76)	0.04 (0.02)	6.9 (0.85)
natural	2.64 (0.08)	37.3 (1.25)	59.75 (1.34)	2.80 (0.48)	0.98 (0.18)	0.72 (0.17)

6.5 DISCUSSION

I found differences in bee capture rates based on landscape and bee species traits. Many of the polylectic bees were social hive nesters (such as *Apis* and *Bombus*) while many of the oligolectic species were solitary ground-nesters (such as *Andrena*). Here, oligolectic and ground-nesting bees here tended to prefer areas with more natural landcover and less non-rewarding agriculture. An important caveat is that bee species traits are not always independent of each other.

I found more than twice as many bees at grazed sites when compared to hay sites. However, this trend is not surprising given that the hay fields contained 99% alfalfa despite the attempt to increase their floral diversity. Alfalfa is attractive to *Bombus* (Wood et al., 2019) and *Megachile* (Pitts-Singer and Cane, 2011), so it is slightly surprising that these genera were not collected in higher abundances at hay sites. Hay sites were dominated by honey bees where other treatments were mostly wild bees. I only identified one other species from the sown hay mix—red clover (*Trifolium pratense* L., 0.047 % $\pm$ 0 observed, 1% of seed mix). Other plants observed were volunteer species that were not included in the seed mix ($n=6$ species); and all had very low abundances ($> 0.4\%$ of ground cover). Given the low abundance of bees at hay sites and the dominant establishment of alfalfa over all other species from the planted seed mix, it is suggested that enhanced hay fields for pollinator conservation should include a higher proportion of other blooming forb species to provide a larger diversity of blooming flowers. In this study A majority (63%) of the seed mix was alfalfa (Table S 7.7), so if conservation is a priority, then low growing Asteraceae species should also be included in the mix that can

withstand regular mowing. Additionally, farmers were instructed to not cut small sections of the crop for hay so pollinators had forage during the entire growing season. Other studies have found this method to be effective and found that pollinator abundance increases over time (Buri et al. 2014). All farmers harvested the entire field in our study. If pollinator conservation is a priority, then small strips of the seed mix should be left for pollinators (Killewald et al. 2024), ideally on the edges of fields where pollinators may be able to find overwintering habitat or may more easily spill over between the fields and adjacent landcover types.

Past research has found that cattle grazing effects on bee diversity and abundance vary, but generally bee populations are inversely related to grazing intensity (Harmon-Threatt and Chin, 2016). Some studies found that the response depends on the group of bees studied, with smaller bees (like sweat bees) being unaffected by grazing and larger (bumble bees) being negatively influenced by grazing practices (Kimoto et al. 2012). A community analysis showed that natural and grazed sites tended to be similar to each other while hay sites were unique. This indicates that grazing an area may support a more natural community of bees than haying the same field. The caveat here is that our sampling took place in July and August, so results may differ if the entire growing season is studied. Though grazed and natural sites were statistically similar, oligolectic species were more abundant in grazed sites whereas polylectic bees were more abundant in natural sites. This could mean that grazing supports more floral specialists than unmanaged natural sites. *Bombus* were most common at natural sites, so they may be causing the slight polylectic bias of natural sites. Andrenidae, mainly *Perdita bruneri* Cockerell ($n=85$) and *Perdita swenki* Crawford ($n=33$) were the most common oligolectic

species and were commonly collected on gumweed (*Grindelia*) found at the grazed sites. In fact, almost every oligolectic bee species I collected specializes on Asteraceae (except *Andrena campanulae*, *A. lupinorum*, and *Colletes andrewsi*). Some species collected, such as *A. campanulae*, are rather rare (8 specimens total in the J.B. Wallis/ R.E. Roughley Museum of Entomology), so collecting even a single specimen indicates that grazed sites support a specialized habitat that is not found in many other locations.

Stem nesting species, mostly Megachilidae, *Hylaeus*, and *Ceratina*, were more common at natural sites, presumably from the increased woody vegetation available here. Hay sites contained no woody vegetation while grazed sites contained some small shrubs (mainly *Symphoricarpos*) that the cattle would avoid. The diversity of woody vegetation was much higher at natural sites as these contained more trees and larger shrub species (M.K. pers. obs.). Possible increases in woody vegetation that could be used as nesting substrate at natural sites may drive these trends, as most of the vegetation variables tested were similar between natural and grazed sites. Hay sites were characterized by low overall capture rates of bees. The bees collected across the study here were mainly hive nesting and polylectic species- but this trend was driven almost exclusively by honey bees. Thus, it is hard to say if hay sites support a functionally distinct community of pollinators outside of the lower overall capture rates.

The diversity of blooming forbs was similar between treatments, but the community structure was different between sites. Grazed sites supported more blooming Asteraceae species, while hay sites were dominated by alfalfa. Thus, it makes sense that Asteraceae specialists would be more abundant within grazed sites. Though individual responses varied, overall, I found more bees in grazed sites when compared to the hay

treatment. This indicates that grazed sites may hold larger populations of bees than monoculture alfalfa hay fields. Both grazed and hay fields are subject to vegetation removal, but the cutting of hay fields creates a mass removal of floral resources in a short period of time, whereas grazing is a slower, more gradual removal of resources. Additionally, grazing may selectively remove vegetation, leaving behind unpalatable species (Milton and Dean, 1990; Mayer, 2004) that may be beneficial for pollinators. Cattle were allowed to graze locations for 7–10 days out of the month and then sites were rested for ~3 weeks post grazing. Given the intensity of grazing at our sites, cattle density was low enough that selective grazing could occur. Cattle tended to not forage on the thistle (*Cirsium*) or snowberry (*Symphoricarpos*) (M.K. Pers. Obs.). Though grazed sites contained more bees than hay sites, natural sites had the greatest abundance of bees. Thus, perennial habitat seems to support larger populations of bees in July and August than managed sites in our study. Nonetheless, in areas of intensive agriculture where converting land to conservation areas may not be possible, utilizing grazing practices instead of haying fields may support higher abundances of pollinators.

Many studies use transects that are placed near site edges (Olynyk et al. 2021) or in easy to access areas (O'Connor et al. 2019). If transects are placed in areas with high floral abundances or that contain visible bees, they will have a higher likelihood of intercepting bees as they forage for resources. If a study aims to determine bee density across the sampling area, then targeting transects in areas with flowers will inflate the calculated population density. Additionally, estimates of bee density per unit area will be non-representative if only the edges of fields are sampled. The edges of fields may contain more floral rewards (that may attract more bees) if plants are able to spill over

from adjacent unmanaged field margins. As expected, targeted transects captured more bees (nearly 50% more), but the effect was only marginally significant ($p=0.06$). I did not identify an interaction between transect type and treatment. So, it appears that the underlying trend may be present regardless of treatment type. We identified that transects placed by the observer may inflate bee population sizes as they did not represent the bee population of the site as a whole (randomly placed transects). Thus, future studies should include randomly placed transects in their study design as it may give a more accurate estimate of insect population density than targeted transects. Depending on the dispersion of bees in the landscape, a stratified sampling approach that includes both randomly and selectively placed transects, similar to that used here, may be necessary to avoid an unreasonable number of transects.

Bees collected in this study did not respond to any of the transect scale variables that I tested with GLMMs but did respond to local variables from the ordination plot. Blooming forb richness, grass cover, and wind speed aggregated bee communities into three different directions. Thus, these variables could be influencing bee populations- but effects are likely weak as they were not significant in the GLMMs. Bees did respond to larger scale variables tested in the GLMMs. Bees were positively associated with natural landcover at 3 km (the best fitting spatial scale), a finding that is supported by previous research as well (Kennedy et al., 2013). Bees can spill over between agricultural and natural landcover types (Montero-Castaño et al. 2016), so it may be possible that bees are using the natural landcover in our study as overwintering habitat or to supplement foraging. The positive association with natural landcover coupled with the natural treatment collecting the highest abundances of bees in our study support the protection of

natural lands for bee conservation. Thus, if possible, landowners should not convert natural lands into agricultural production if bee conservation is a priority.

I found a positive association between field size and bee abundances among the three treatments, meaning that larger fields generally contained more bees. This finding was present regardless of treatment. It should be noted that hay sites showed a slight negative relationship between bee abundances and field size, but the hay fields were all small (< 100 ha in size) compared to grazed and natural sites, so it is unclear if this relationship will hold true for larger hay fields. It may be possible that larger fields have more areas for cattle to graze, reducing overall grazing intensity at the field level. I rejected our hypothesis that larger fields may provide more opportunities to collect bees in our targeted transects as I found no interaction between transect type and field size and transect type marginally influenced bee capture rates overall.

6.6 CONCLUSION

Historically, prairies covered a large area of central North America and were home to many insects. Today, insects must find ways to survive within these shrinking habitats. Many forms of agriculture are detrimental to insect populations, but livestock grazing may provide an opportunity to cultivate a managed grassland that is beneficial for both livestock and insects. Here, I found that grazing practices increased abundances of blooming Asteraceae and their associated pollinator specialists in July and August. Even though bee abundances were statistically similar between grazed and natural sites, oligolectic bees were more abundant within grazed fields, but polylectic bees were most abundant within natural sites. Natural sites were characterized by high collections of stem

nesters and *Bombus*. Hay fields in July and August were characterized by high amounts of alfalfa and no functional group of bees were associated with these sites. These hay fields were sampled before or between cuttings in times when flowers were in bloom. Though *A. mellifera* was the most abundant species from hay sites. Grazed sites contained more bees than hayed sites, more bees were collected at natural sites than any of the other two treatments. This indicates that although grazing practices likely support larger populations of pollinators than hayed monoculture fields, natural lands support larger populations of bees. Additionally, bee capture rates were positively associated with natural land cover. Thus, land should be set aside in areas with high agricultural intensity in order to preserve the ecosystem services offered by beneficial insects.

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CHAPTER 7. GENERAL DISCUSSION

7.1 INTRODUCTION TO GENERAL DISCUSSION

Insect ecosystem services help to maintain agricultural (Zhang et al. 2007) and natural (Ollerton et al. 2011) systems across the globe. This is problematic because insect communities are often less diverse, populated, and / or resilient in agriculturally dominated landscapes (Gallai et al., 2009; Wagner et al., 2021; Mancini et al., 2023). Policy change has aimed to mitigate insect declines, but this has not proven to be effective enough as declines continue even in protected areas (Mancini et al. 2023). Additional measures have been taken by landowners to support insect communities and these have shown promising results. Habitat enhancements placed within or near agricultural fields provide areas where insects can find food and overwintering locations (Langellotto and Denno, 2004). These enhancements support insect communities (Ganser et al. 2021), but results often vary between study locations and landscape context (Karp et al. 2018). I studied the use of habitat enhancements and local farming practices to support beneficial insects and monitored insect responses for multiple years following establishment of floral enhancements.

7.2 THESIS SUMMARY

I installed floral strips adjacent to annual crop fields in Manitoba, Canada in the summer of 2019 and then monitored beneficial insect communities for four field seasons. I analysed insect population density, diversity, and structure to determine how insect communities change following the installation of the floral strips. Additionally, I

measured landscape and local environmental variables at each site to determine how landscape context effects the interaction between insects and floral strips.

I also surveyed enhanced hay fields and compared those to grazed pastures and natural sites to determine the effects of a floral enhancement that was managed for agricultural production. Hay fields were planted with a pollinator enhanced seed mix in 2018 and were cut and bailed for hay twice each summer. Grazed sites were rotationally grazed by cattle with roughly one week on pasture and three weeks off pasture. Natural sites were not managed (not cut for hay or grazed) for at least 5 years prior to sampling. I measured insect population density, diversity, and functional responses to treatment type (hay, grazed, or natural). I also investigated landscape context and environmental variables to determine relationships between insects and their surrounding habitat. Insect responses to each management regime are summarized below. Additionally, recommendations for next steps are given and their importance is discussed.

7.2.1 Beneficial insect numerical responses to floral strips

I found that the beneficial insects I studied had varying responses to the addition of floral strips. Responses not only varied between taxa but also years. As insect collection methods changed after the first field season, I cannot directly compare measurements from the first year (2019) and subsequent years (2020–2022). General trends for each taxon are discussed below.

7.2.1.1 *Ground beetles (Coleoptera: Carabidae)*

Ground beetle capture rates in year one were three times greater than the control and nine times greater than the natural sites. Diversity was similar between treatments in year 1. Carabid capture rates and diversity was highest in floral strips in year 2 but steeply declined in year three to similar levels as control sites. Natural sites captured the lowest abundance of beetles across all sampling years while control sites were mostly intermediate between strip and natural sites. Our results are supported by other literature on beetle banks that show increased beetle capture rates following habitat manipulations (Thomas, 2001).

7.2.1.2 *Hover flies (Diptera: Syrphidae)*

Syrphid capture rates or diversity never responded positively to the addition of floral strips. In fact, their capture rates were higher within the grassy margins of the control fields in the first year of establishment. Syrphid capture rates were similar between treatments during years 2–4. The majority of our syrphids were *Toxomerus*, so the individual preferences of this taxon drove the pattern I observed. It is still unclear why syrphids responded this way. What is clear is that the addition of floral strips did not positively influence this taxon in this study.

7.2.1.3 *Bees (Hymenoptera: Apoidea)*

Bee capture rates were four and 24 times more abundant in strips than control fields when collected with pan traps and nets, respectively. Though diversity was similar

between treatments in year 1 for pan traps, bees collected with nets were more diverse in strip sites compared to control. I did not identify an overall effect of treatment across all sampling methods (pan trap, blue vane trap, and net transects) for bees collected in years 2–4. However, bees were twice as abundant in strip sites when compared to control and natural sites. Capture rates declined slightly in year 4 compared to the previous years, which could be partially due to the flooding conditions observed that same summer. Nonetheless, strips seem effective at increasing the population sizes of bees and their capture rates increased over time. These results are supported by a large body of literature showing that habitat enhancements support bees (Blaauw and Isaacs, 2014; Kordbach et al. 2020; Albrecht *et al.*, 2021) and other beneficial insects.

7.2.2 Spillover of beneficial insects into adjacent fields

I found evidence that bees spillover from floral strips to adjacent fields both during canola bloom and during the entire summer. Bee abundance and diversity was higher in crops adjacent to the strips when compared to a grassy margin of the same field—regardless of distance within the adjacent field (measured up to 50 m). Other studies on natural enemies have found that edge effects may extend up to 200 m into fields (Morandin et al. 2014), but it is unclear how much farther the edge effects extend in this study since 50 m was the extent of our observations. This indicates that floral strips are effective at increasing populations of bees internally and that these populations will forage outside of the strips and into the fields for resources. However, more research is necessary to determine effects of bee spillover into adjacent crops and more specifically, if crop yield responds to the increases in bee populations.

I did not find evidence of spillover for ground beetles or hover flies in our study. Capture rates and diversity of hover flies were similar between most of the sampled areas. However, ground beetle capture rates within the field adjacent to the strips (15 m) were lower than both the strips and further within the field (50 m). Though this difference was not statistically significant, a general decrease in carabid capture rates can be observed (Figure 4.3A) and may indicate that the floral strips are acting as a sink of carabids rather than a source (Chen, Fu and Zhao, 2008). Other literature finds that ground beetles may use these enhancements for locomotion (Alignier et al., 2020). Thus, these floral strips could aid in the dispersal of ground dwelling insects between possibly hostile landcover types.

7.2.3 Landscape

Ground beetles always responded strongest at the 500 m spatial scale, indicating that they respond to their environment at relatively small scales. Ground beetles responded positively to areas with less habitat fragmentation. Carabids have been found to use certain landcover types to move within the landscape (Charrier, Petit and Burel, 1997), so they could use the floral strips as a corridor to move between habitats. However, the literature on carabid dispersal is not well documented so further research should be completed to understand to what degree ground beetles and other ground dwelling insects may use these corridors for locomotion.

Ground beetles responded negatively to grass cover in year one but then responded positively in years two and three. Pitfall traps measure activity density of ground predators and as such, capture rates may change based on the surrounding

vegetation. Vegetation density is negatively correlated with capture rates, so some authors (Greenslade 1964) have created a standardized patch of bare vegetation to compare treatment types. Since I did not follow this same protocol, I cannot separate if differences in vegetation structure between treatments is due to carabid preference or difficulty of movement.

Both syrphid abundance and diversity was negatively associated with edge density (500 m). Syrphid abundance was also positively associated with landscape diversity (500 m). This indicates that syrphids may prefer areas with larger patches of habitat within the immediate landscape.

Bee capture rates tended to be higher in locations with increased forb cover, higher Shannon landscape diversity, and less developed lands within the broad landscape (3 kilometres (km)). Bee diversity was influenced at a smaller (1 km) scale than abundance, but diversity was not associated with any tested variables here. These results are supported by previous studies as well that show bees are positively associated with natural landcover and habitat diversity within the landscape (Aguilera et al. 2020; Ratto et al. 2021; Hevia et al. 2021).

7.2.4 Beneficial insect reproduction

Floral strips increased nesting rates, number of offspring per nest, and dietary diversity compared to other treatments of stem-nesting leafcutter bees (*Megachile*). In fact, of the 122 nests collected, only three of them were from control fields. Studies on the nesting habits of bees adjacent to floral strips are rare, but a similar study found the same trends—where only a single stem was collected from control fields (Graham et al.

2020). *Megachile rotundata* is commonly managed for alfalfa seed production (Pitts-Singer and Cane, 2011) and was collected here. The addition of floral strips may help to support population growth and stability of this managed pollinator- but further research is necessary to determine if the strips compete with the crop for pollinators.

I found that the pollen diversity within stems at floral strip sites was highest of all the treatments and that bees selectively collected pollen as the communities of plants and collected pollens were distinct from each other. In our study, nesting rates at strip sites were similar to natural sites. This result is promising given that stem-nesting bee species were significantly associated with natural sites. This indicates that strips are effective at supporting populations of stem-nesting bees and may support reproduction of this group more so than the natural sites they are associated with. I only tested reproductive potential on *Megachile*, but the effects observed here should be broadly applicable across bee taxa. Most bees nest in the ground, but due to the challenges associated with locating and excavating underground nests, stem-nesting bees provide a useful proxy for the bee community. Given that limited dietary diversity can cause population declines of bees (Wood et al. 2019), strips seem to be an important tool to increase the available nutritional resources that bees require.

7.2.5 Floral strip establishment

I found that insect responses changed over the years, and this may be in part due to changes in vegetation structure. Forb cover decreased and grass cover increased as the strips established, indicating that, at least four years after establishment, the strips did reach an equilibrium. Other studies have found that it may take up to four years for insect

communities to respond to the addition of floral strips (Blaauw and Isaacs, 2014), so it may be possible that the same (or longer) length of time is necessary for the strips themselves to establish. There is no doubt that weather fluctuations observed, both drought and flooding conditions, hindered strip establishment, but to what extent is unclear. I observed that several of the strips failed to establish, with two strips never establishing and others failing after a year or two.

The seed mix used in our study was rather novel, with the inclusion of annuals intended to provide ample flowers and ground cover during the first year of establishment to suppress weeds. Though the annuals established well in year 1 (M.K. pers. obs.) the establishment of the perennial species was rather low overall (years 1+). This could be in part due to the low seeding rate of the perennials, seeding method (broadcast seeding), or due to the soil conditions. During the study period, drought conditions persisted causing the soil to dry and form a surface crust (M.K. pers. obs), so germination of broadcast seed in this study was particularly low. It is possible that irrigation of the strips or utilizing a drill seeder to provide direct seed to soil contact would have improved establishment. Additionally, the strips that did not establish in the first year of the study were located in fields with reduced tillage practices, that likely reduce seed to soil contact and thus establishment. Sites that utilized conventional tillage practices generally had better strip establishment, and this observation should be studied further. Studying strip establishment in multiple soil contexts is important as tillage system, residue management, soil type, and drainage class likely effect plant establishment and strip longevity.

7.2.6 Pasture management

I found that bees were equally as abundant within grazed pastures and natural sites in July and August. Both treatments contained more bees than hay sites. Additionally, hay sites contained almost exclusively honey bees. Though it is important to note that the hay fields in this study established almost exclusively alfalfa (99 %)- so differences in bee capture rates here are likely tied to the floral communities at these sites. This indicates that grazing, rather than haying fields, may help to conserve local bee populations. Though grazed and natural sites supported similar numbers of bees, the community structure was different. Grazed sites supported more floral specialist species—oligolectic bees that were mainly Andrenidae—whereas natural sites supported more stem-nesting species. Thus, it seems that the addition of grazing practices may help to increase the diversity of niches in the overall landscape and support landscape level biodiversity.

The act of cutting forage fields for hay rather than grazing was not the source of the difference between treatments. The hay sites sampled here were almost exclusively alfalfa. Other species were included in the seed mix (Table S 7.7), but few, if any, other species were observed in the plantings. Thus, if conservation of insects is a priority, then the diversity of the forage seed mix should be increased or alter the management style to support more flowering species in both pastures and hay fields. Additionally, farmers were instructed to leave a strip of uncut vegetation as a refuge for beneficial insects, though this almost never happened in practice. Leaving a strip of vegetation has proven effective in other studies (Buri et al. 2014) and it is unfortunate that I could not test this practice in this study. Nonetheless, the practice of leaving a refuge for insects is

encouraged for future management studies. Grazed site management inherently include this practice, as rotational razing was practiced and allowed livestock to graze a paddock within a field for one week followed by a rested for three weeks. This rest period allowed flowers to regrow and the bees a chance to forage within the field following grazing.

7.3 FINAL CONCLUSIONS AND RECCOMENDATIONS

Floral strips have been shown to be an effective management and conservation tool for some groups of beneficial insects in Manitoba, Canada. Though syrphids did not respond to the floral strips, ground beetles and bees were both positively affected by their addition to the landscape. Bees spilled over into the adjacent fields, which may ultimately increase the ecosystem services that they provide to crops. At least one group of bees, which included the managed species, *Megachile rotundata*, preferentially nested near floral strips. I acknowledge that installation of floral strips may be difficult and time consuming. If this is not possible, then leaving land unmanaged may be a good approach as natural sites in our study supported beneficial insects, so increasing their abundance within the landscape should be a priority.

7.4 FUTURE DIRECTIONS

Beneficial insect populations continue to decline (Mancini et al. 2023) and there is no doubt that intervention is necessary to improve conservation efforts. To date, almost all studies on floral enhancements have been located within the northern hemisphere, with an emphasis on the United States and Europe (Zamorano et al. 2020). Researching

the effects of enhancements from other areas of the world is important for global conservation of arthropod mediated ecosystem services. Studies should include not only insect responses to the enhancements, but responses of the enhancements to the environment. This can and should include soil type, hydrology, seeding date and rate, and other factors that may affect germination and overall establishment. Floral strips and other enhancements seem to be an effective management and conservation tool for some groups of beneficial insects, but their implementation needs to be refined in additional areas if global insect conservation is to be achieved.

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SUPPLEMENTARY TABLES

Table S 7.1 Composition of the seed mix used to establish the floral strips. Percentages are based on the seed weight. Seed was tested for the presence of foreign species prior to mixing.

% of mix	Family	Genus	species	common name
34.444	Polygonaceae	<i>Fagopyrum</i>	<i>esculentum</i>	Buckwheat
26.526	Fabaceae	<i>Onobrychis</i>	<i>viciifolia</i>	Common Sainfoin
11.217	Poaceae	<i>Thinopyrum</i>	<i>intermedium</i>	Intermediate Wheatgrass
5.675	Asteraceae	<i>Helianthus</i>	<i>annuus</i>	Common Sunflower
3.563	Boraginaceae	<i>Phacelia</i>	<i>tanacetifolia</i>	Lacy Phacelia
2.640	Fabaceae	<i>Trifolium</i>	<i>incarnatum</i>	Crimson Clover
2.587	Fabaceae	<i>Dalea</i>	<i>candida</i>	White Prairie Clover
2.560	Fabaceae	<i>Dalea</i>	<i>purpurea</i>	Purple Prairie Clover
2.362	Poaceae	<i>Nassella</i>	<i>viridula</i>	Green Needlegrass
1.978	Poaceae	<i>Schizachyrium</i>	<i>scoparium</i>	Little Bluestem
1.584	Fabaceae	<i>Medicago</i>	<i>sativa</i>	Alfalfa
1.584	Asteraceae	<i>Cichorium</i>	<i>intybus</i>	Common Chicory
1.584	Poaceae	<i>Bouteloua</i>	<i>gracilis</i>	Blue Grama
0.660	Fabaceae	<i>Trifolium</i>	<i>resupinatum</i>	Persian Clover
0.266	Fabaceae	<i>Trifolium</i>	<i>pratense</i>	Red Clover
0.264	Fabaceae	<i>Trifolium</i>	<i>hybridum</i>	Alsike Clover
0.090	Asteraceae	<i>Helianthus</i>	<i>pauciflorus</i>	Stiff Sunflower
0.022	Apiaceae	<i>Zizia</i>	<i>aptera</i>	Heartleaf Golden Alexander
0.001	Asteraceae	<i>Solidago</i>	<i>nemoralis</i>	Gray Goldenrod

Table S 7.2 Average percent ground cover and standard error (in parenthesis) of each plant species identified from the vegetation surveys across the three treatment groups. Data were collected in the summer of 2019. Treatments included strip, locations adjacent to crops with floral enhancement; control, locations adjacent to crops without any enhancements; and natural, semi-natural areas without management.

species	strip	control	natural
<i>Agalinis aspera</i>	0	0	0.01 (0.01)
<i>Apocynum androsaemifolium</i>	0	0	0.02 (0.02)
<i>Brassica sp</i>	0.1 (0.06)	0.01 (0.01)	0
<i>Cichorium intybus</i>	0	0	0.02 (0.01)
<i>Cirsium arvense</i>	0.18 (0.1)	1.04 (0.61)	1.34 (0.59)
<i>Dasiphora fruticosa</i>	0	0	0.51 (0.51)
<i>Euthamia graminifolia</i>	0	0	0.23 (0.12)
<i>Fagopyrum esculentum</i>	18.63 (2.64)	0	0
<i>Gentianopsis virgata</i>	0	0	0.02 (0.02)
<i>Glycyrrhiza lepidota</i>	0	0	0.26 (0.18)
<i>Helianthus annuus</i>	2.33 (0.72)	0	0
<i>Linum usitatissimum</i>	0.01 (0.01)	0	0
<i>Lotus corniculatus</i>	0	0	1.51 (0.94)
<i>Medicago lupulina</i>	0	0.02 (0.02)	0
<i>Medicago sativa</i>	0	2.35 (0.92)	0
<i>Melilotus albus</i>	0.14 (0.14)	0.22 (0.22)	0
<i>Melilotus officinalis</i>	0	0.64 (0.34)	0
<i>Persicaria maculosa</i>	0	0	0.07 (0.04)
<i>Phacelia tanacetifolia</i>	6.51 (1.13)	0	0
<i>Solidago canadensis</i>	0	0.09 (0.08)	0.21 (0.12)
<i>Sonchus asper</i>	2.21 (0.71)	0.02 (0.02)	0
<i>Symphyotrichum oolentangiense</i>	0	0	0.02 (0.02)
<i>Trifolium incarnatum</i>	0.23 (0.15)	0	0
<i>Trifolium pratense</i>	0.14 (0.07)	0.1 (0.06)	0
<i>Trifolium repens</i>	0.78 (0.4)	0.01 (0.01)	0.74 (0.55)
<i>Vicia cracca</i>	1.1 (0.49)	0	0

Table S 7.3 Counts of each species collected in each sampling method across all sites sampled in 2019.

Method	Family	Genus	Species	Author	n
pitfall trap	Carabidae	<i>Agonum</i>	<i>cupreum</i>	Dejean 1831	319
pitfall trap	Carabidae	<i>Agonum</i>	<i>cupripenne</i>	Say 1823	8
pitfall trap	Carabidae	<i>Agonum</i>	<i>gratiosum</i>	Mannerheim 1853	3
pitfall trap	Carabidae	<i>Agonum</i>	<i>placidum</i>	Say 1823	482
pitfall trap	Carabidae	<i>Amara</i>	<i>apricaria</i>	(Paykull 1790)	2
pitfall trap	Carabidae	<i>Amara</i>	<i>aulica</i>	(Panzer 1796)	10
pitfall trap	Carabidae	<i>Amara</i>	<i>aulica</i> sp.group		4
pitfall trap	Carabidae	<i>Amara</i>	<i>carinata</i>	(LeConte 1847)	4
pitfall trap	Carabidae	<i>Amara</i>	<i>cupreolata</i>	Putzeys 1866	7
pitfall trap	Carabidae	<i>Amara</i>	<i>ellipsis</i>	(Casey 1918)	1
pitfall trap	Carabidae	<i>Amara</i>	<i>erratica</i>	(Duftschmidt 1812)	1
pitfall trap	Carabidae	<i>Amara</i>	<i>farcta</i>	LeConte 1855	221
pitfall trap	Carabidae	<i>Amara</i>	<i>impuncticollis</i>	(Say 1823)	68
pitfall trap	Carabidae	<i>Amara</i>	<i>littoralis</i>	Mannerheim 1843	665
pitfall trap	Carabidae	<i>Amara</i>	<i>obesa</i>	Say 1823	22
pitfall trap	Carabidae	<i>Amara</i>	<i>pallipes</i>	Kirby 1837	3
pitfall trap	Carabidae	<i>Amara</i>	<i>quenseli</i>	(Schonherr 1806)	4
pitfall trap	Carabidae	<i>Amara</i>	<i>sp.</i>		1
pitfall trap	Carabidae	<i>Amara</i>	<i>torrida</i>	(Panzer 1796)	3
pitfall trap	Carabidae	<i>Anisodactylus</i>	<i>nigrita</i>	Dejean 1829	1
pitfall trap	Carabidae	<i>Anisodactylus</i>	<i>sanctaecrucis</i>	(Fabricius 1798)	39
pitfall trap	Carabidae	<i>Badister</i>	<i>obtusus</i>	LeConte 1878	4
pitfall trap	Carabidae	<i>Bembidion</i>	<i>nitidum</i>	(Kirby 1837)	4
pitfall trap	Carabidae	<i>Bembidion</i>	<i>obscurum</i>	(Motschulsky 1845)	52
pitfall trap	Carabidae	<i>Bembidion</i>	<i>quadrimaculatum</i>	(Linnaeus 1761)	681
pitfall trap	Carabidae	<i>Bembidion</i>	<i>rapidum</i>	(LeConte 1847)	10
pitfall trap	Carabidae	<i>Bembidion</i>	<i>rupicola</i>	(Kirby 1837)	25
pitfall trap	Carabidae	<i>Bembidion</i>	<i>sp.</i>		7
pitfall trap	Carabidae	<i>Bembidion</i>	<i>versicolor</i>	(LeConte 1848)	10
pitfall trap	Carabidae	<i>Brachinus</i>	<i>quadrapennis</i>	Dejean 1825	1
pitfall trap	Carabidae	<i>Bradycellus</i>	<i>congener</i>	(LeConte 1847)	1
pitfall trap	Carabidae	<i>Bradycellus</i>	<i>neglectus</i>	(LeConte 1847)	2
pitfall trap	Carabidae	<i>Calleida</i>	<i>punctata</i>	LeConte 1848	1
pitfall trap	Carabidae	<i>Calosoma</i>	<i>calidum</i>	(Fabricius 1775)	107
pitfall trap	Carabidae	<i>Calosoma</i>	<i>lepidum</i>	LaConte 1844	1

Table S 7.3 cont'd

Method	Family	Genus	Species	Author	n
pitfall trap	Carabidae	<i>Carabus</i>	<i>granulatus</i>	Linnaeus 1758	13
pitfall trap	Carabidae	<i>Chlaenius</i>	<i>alternatus</i>	Horn 1871	1
pitfall trap	Carabidae	<i>Chlaenius</i>	<i>purporicollis</i>	Randall 1838	2
pitfall trap	Carabidae	<i>Chlaenius</i>	<i>sericeus</i>	(Forster 1771)	2
pitfall trap	Carabidae	<i>Cymindis</i>	<i>neglecta</i>	Haldeman 1843	16
pitfall trap	Carabidae	<i>Dicaelus</i>	<i>sculptilis</i>	Say 1823	37
pitfall trap	Carabidae	<i>Diplocheila</i>	<i>assimilis</i>	(LeConte 1844)	1
pitfall trap	Carabidae	<i>Diplocheila</i>	<i>obtusa</i>	(LeConte 1848)	9
pitfall trap	Carabidae	<i>Dyschirius</i>	<i>globulosus</i>	(Say 1823)	4
pitfall trap	Carabidae	<i>Elaphropus</i>	<i>incurvus</i>	(Say 1830)	1
pitfall trap	Carabidae	<i>Harpalus</i>	<i>affinis</i>	(Schränk 1781)	14
pitfall trap	Carabidae	<i>Harpalus</i>	<i>amputatus</i>	Say 1830	3
pitfall trap	Carabidae	<i>Harpalus</i>	<i>eraticus</i>	Say 1823	122
pitfall trap	Carabidae	<i>Harpalus</i>	<i>erythropus</i>	Dejean 1829	1
pitfall trap	Carabidae	<i>Harpalus</i>	<i>laevipes</i>	Zetterstedt 1828	2
pitfall trap	Carabidae	<i>Harpalus</i>	<i>lewisi</i>	LeConte 1865	3
pitfall trap	Carabidae	<i>Harpalus</i>	<i>opacipennis</i>	(Haldeman 1843)	30
pitfall trap	Carabidae	<i>Harpalus</i>	<i>pensylvanicus</i>	(DeGeer 1774)	267
pitfall trap	Carabidae	<i>Harpalus</i>	<i>ventralis</i>	LeConte 1848	4
pitfall trap	Carabidae	<i>Loricera</i>	<i>pilicornis</i>	(Fabricius 1775)	1
pitfall trap	Carabidae	<i>Microlestes</i>	<i>linearis</i>	(LeConte 1851)	25
pitfall trap	Carabidae	<i>Oxypselaphus</i>	<i>pusillus</i>	(LeConte 1854)	17
pitfall trap	Carabidae	<i>Poecilus</i>	<i>corvus</i>	(LeConte 1873)	70
pitfall trap	Carabidae	<i>Poecilus</i>	<i>lucublandus</i>	(Say 1823)	139
pitfall trap	Carabidae	<i>Pterostichus</i>	<i>adstrictus</i>	Eschscholtz 1823	1
pitfall trap	Carabidae	<i>Pterostichus</i>	<i>commutabilis</i>	(Motschulsky 1866)	3
pitfall trap	Carabidae	<i>Pterostichus</i>	<i>corvinus</i>	(Dejean 1828)	3
pitfall trap	Carabidae	<i>Pterostichus</i>	<i>femoralis</i>	(Kirby 1837)	5
pitfall trap	Carabidae	<i>Pterostichus</i>	<i>melanarius</i>	(Illiger 1798)	525
pitfall trap	Carabidae	<i>Pterostichus</i>	<i>novus</i>	Straneo 1944	1
pitfall trap	Carabidae	<i>Stenolophus</i>	<i>comma</i>	(Fabricius 1775)	18
pitfall trap	Carabidae	<i>Stenolophus</i>	<i>lineola</i>	(Fabricius 1775)	4
pitfall trap	Carabidae	<i>Synuchus</i>	<i>impunctatus</i>	(Say 1823)	5

Table S 7.3 cont'd

Method	Family	Genus	Species	Author	n
pan trap	Syrphidae	<i>Eristalis</i>	<i>stipator</i>	Osten Sacken 1877	6
pan trap	Syrphidae	<i>Eristalis</i>	<i>tenax</i>	(Linnaeus 1758)	6
pan trap	Syrphidae	<i>Eupeodes</i>	<i>americanus</i>	(Wiedemann 1830)	28
pan trap	Syrphidae	<i>Eupeodes</i>	<i>volucris</i>	Osten Sacken 1877	1
pan trap	Syrphidae	<i>Helophilus</i>	<i>hybridus</i>	Loew 1846	2
pan trap	Syrphidae	<i>Helophilus</i>	<i>latifrons</i>	Loew 1863	10
pan trap	Syrphidae	<i>Helophilus</i>	<i>obscurus</i>	Loew 1863	1
pan trap	Syrphidae	<i>Parhelophilus</i>	<i>obsoletus</i>	(Loew 1863)	1
pan trap	Syrphidae	<i>Platycheirus</i>	sp.		5
pan trap	Syrphidae	<i>Sphaerophoria</i>	<i>contigua</i>	Macquart 1847	2
pan trap	Syrphidae	<i>Sphaerophoria</i>	<i>philanthus</i>	(Meigen 1822)	5
pan trap	Syrphidae	<i>Sphaerophoria</i>	<i>pyrrhina</i>	Bigot 1884	2
pan trap	Syrphidae	<i>Syritta</i>	<i>pipiens</i>	(Linnaeus 1758)	1
pan trap	Syrphidae	<i>Syrphus</i>	<i>rectus</i>	Osten Sacken 1875	1
pan trap	Syrphidae	<i>Syrphus</i>	<i>ribesii</i>	(Linnaeus 1758)	1
pan trap	Syrphidae	<i>Toxomerus</i>	<i>geminatus</i>	Say 1823	33
pan trap	Syrphidae	<i>Toxomerus</i>	<i>marginatus</i>	Say 1823	1110
net	Andrenidae	<i>Andrena</i>	<i>lupinorum</i>	Cockerell 1906	1
net	Andrenidae	<i>Andrena</i>	<i>miranda</i>	Smith 1879	1
net	Andrenidae	<i>Andrena</i>	<i>prunorum</i>	Cockerell 1896	1
net	Andrenidae	<i>Andrena</i>	<i>thaspiae</i>	Graenicher 1903	1
net	Andrenidae	<i>Protandrena</i>	<i>perlaevis</i>	(Swenk and Cockerell 1907)	1
net	Apidae	<i>Anthophora</i>	<i>terminalis</i>	Cresson 1869	3
net	Apidae	<i>Apis</i>	<i>mellifera</i>	Linnaeus 1758	266
net	Apidae	<i>Bombus</i>	<i>bimaculatus</i>	Cresson 1863	18
net	Apidae	<i>Bombus</i>	<i>borealis</i>	Kirby 1837	21
net	Apidae	<i>Bombus</i>	<i>griseocollis</i>	(De Geer 1773)	19
net	Apidae	<i>Bombus</i>	<i>huntii</i>	Greene 1860	1
net	Apidae	<i>Bombus</i>	<i>nevadensis</i>	Cresson 1874	8
net	Apidae	<i>Bombus</i>	<i>rufocinctus</i>	Cresson 1863	43
net	Apidae	<i>Bombus</i>	<i>ternarius</i>	Say 1837	95
net	Apidae	<i>Bombus</i>	<i>terricola</i>	Kirby 1837	5
net	Apidae	<i>Bombus</i>	<i>vagans</i>	Smith 1854	29
net	Apidae	<i>Ceratina</i>	<i>calcarata</i>	Robertson 1900	1
net	Apidae	<i>Ceratina</i>	<i>mikmaqi</i>	Rehan and Sheffield 2011	1
net	Apidae	<i>Melissodes</i>	<i>agilis</i>	Cresson 1878	43
net	Apidae	<i>Melissodes</i>	<i>confusus</i>	Cresson 1878	1
net	Apidae	<i>Melissodes</i>	<i>perlusus</i>	Cockerell 1925	1
net	Apidae	<i>Melissodes</i>	<i>trinodis</i>	Robertson 1901	16
net	Apidae	<i>Triepeolus</i>	<i>helianthi</i>	(Robertson 1897)	1

Table S 7.3 cont'd

Method	Family	Genus	Species	Author	n
net	Apidae	<i>Triepeolus</i>	<i>obliteratus</i>	Graenicher 1911	1
net	Colletidae	<i>Colletes</i>	<i>sp. (Hyalinus?)</i>		1
net	Colletidae	<i>Hylaeus</i>	<i>affinis</i>	(Smith 1853)	6
net	Colletidae	<i>Hylaeus</i>	<i>annulatus</i>	(Linnaeus 1758)	1
net	Colletidae	<i>Hylaeus</i>	<i>mesillae</i>	(Cockerell 1896)	1
net	Halictidae	<i>Halictus</i>	<i>confusus</i>	Smith 1853	3
net	Halictidae	<i>Lasioglossum</i>	<i>laevissimum</i>	(Smith 1853)	1
net	Halictidae	<i>Lasioglossum</i>	<i>leucozonium</i>	(Schrank 1781)	1
net	Halictidae	<i>Lasioglossum</i>	<i>zonulus</i>	(Smith 1848)	1
net	Megachilidae	<i>Hoplitis</i>	<i>producta</i>	(Cresson 1864)	3
net	Megachilidae	<i>Megachile</i>	<i>inermis</i>	Provancher 1888	1
net	Megachilidae	<i>Megachile</i>	<i>latimanus</i>	Say 1823	7
net	Megachilidae	<i>Megachile</i>	<i>relativa</i>	Cresson 1878	1
pan trap	Andrenidae	<i>Andrena</i>	<i>chromotricha</i>	Cockerell 1899	2
pan trap	Andrenidae	<i>Andrena</i>	<i>thaspii</i>	Graenicher 1903	47
pan trap	Andrenidae	<i>Protandrena</i>	<i>parvus</i>	(Robertson 1892)	1
pan trap	Andrenidae	<i>Protandrena</i>	<i>perlaevis</i>	(Swenk and Cockerell 1907)	1
pan trap	Apidae	<i>Anthophora</i>	<i>terminalis</i>	Cresson 1869	1
pan trap	Apidae	<i>Apis</i>	<i>mellifera</i>	Linnaeus 1758	68
pan trap	Apidae	<i>Bombus</i>	<i>borealis</i>	Kirby 1837	14
pan trap	Apidae	<i>Bombus</i>	<i>griseocollis</i>	(De Geer 1773)	3
pan trap	Apidae	<i>Bombus</i>	<i>rufocinctus</i>	Cresson 1863	4
pan trap	Apidae	<i>Bombus</i>	<i>ternarius</i>	Say 1837	33
pan trap	Apidae	<i>Bombus</i>	<i>terricola</i>	Kirby 1837	1
pan trap	Apidae	<i>Bombus</i>	<i>vagans</i>	Smith 1854	7
pan trap	Apidae	<i>Melissodes</i>	<i>agilis</i>	Cresson 1878	54
pan trap	Apidae	<i>Melissodes</i>	<i>bimaculatus</i>	(Lepeletier 1825)	4
pan trap	Apidae	<i>Melissodes</i>	<i>confusus</i>	Cresson 1878	11
pan trap	Apidae	<i>Melissodes</i>	<i>trinodis</i>	Robertson 1901	20
pan trap	Apidae	<i>Triepeolus</i>	<i>obliteratus</i>	Graenicher 1911	1
pan trap	Colletidae	<i>Hylaeus</i>	<i>affinis</i>	(Smith 1853)	2
pan trap	Colletidae	<i>Hylaeus</i>	<i>annulatus</i>	(Linnaeus 1758)	1
pan trap	Halictidae	<i>Halictus</i>	<i>confusus</i>	Smith 1853	7
pan trap	Halictidae	<i>Halictus</i>	<i>rubicundus</i>	Christ 1791	5
pan trap	Halictidae	<i>Lasioglossum</i>	<i>admirandum</i>	(Sandhouse 1924)	1
pan trap	Halictidae	<i>Lasioglossum</i>	<i>albipenne</i>	(Robertson 1890)	17
pan trap	Halictidae	<i>Lasioglossum</i>	<i>athabascense</i>	(Sandhouse 1933)	1
pan trap	Halictidae	<i>Lasioglossum</i>	<i>cinctipes</i>	(Provancher 1888)	4

Table S 7.3 cont'd

Method	Family	Genus	Species	Author	n
pan trap	Halictidae	<i>Lasioglossum</i>	<i>coriaceum</i>	(Smith 1853)	1
pan trap	Halictidae	<i>Lasioglossum</i>	<i>leucozonium</i>	(Schrank 1781)	1
pan trap	Halictidae	<i>Lasioglossum</i>	<i>novascotiae</i>	(Mitchell 1960)	2
pan trap	Halictidae	<i>Lasioglossum</i>	<i>paraforbesii</i>	McGinley 1986	10
pan trap	Halictidae	<i>Lasioglossum</i>	<i>planatum</i>	(Lovell 1905)	2
pan trap	Halictidae	<i>Lasioglossum</i>	<i>sagax</i>	(Sandhouse 1924)	2
pan trap	Halictidae	<i>Lasioglossum</i>	<i>semicaeruleum</i>	(Cockerell 1895)	3
pan trap	Halictidae	<i>Lasioglossum</i>	<i>zephyrus</i>	(Smith 1853)	6
pan trap	Halictidae	<i>Lasioglossum</i>	<i>zonulus</i>	(Smith 1848)	1
pan trap	Megachilidae	<i>Heriades</i>	<i>carinata</i>	Cresson 1864	3
pan trap	Megachilidae	<i>Megachile</i>	<i>inermis</i>	Provancher 1888	3
pan trap	Megachilidae	<i>Megachile</i>	<i>latimanus</i>	Say 1823	4

Table S 7.4 Average (standard error) forb species richness and ground cover of forbs grasses and bare ground that was identified from the vegetation surveys from 2020–2022. Data were collected in Manitoba Canada. Treatments included strip locations adjacent to crops with floral strips; control locations adjacent to crops without any enhancements; and semi-natural areas without management. *n* values indicate the total number of transects summarized for each variable with eleven 0.5 m square quadrats measured for each individual transect. Values given are mean ($\pm$ SE) of forb richness and percent ground cover of forbs grasses and bare ground per quadrat.

Treatment	Year	<i>n</i>	Forb richness	Forb cover	Grass cover	Bare cover
control	2020	43	0.27 (0.06)	14.48 (2.86)	84.09 (3.19)	1.43 (0.76)
	2021	50	0.21 (0.03)	16.03 (2.18)	82.41 (2.28)	1.56 (0.64)
	2022	18	0.36 (0.08)	15.33 (3.03)	84.47 (3.07)	0.20 (0.16)
natural	2020	46	0.38 (0.06)	27.11 (3.25)	72.76 (3.31)	0.13 (0.13)
	2021	50	0.33 (0.06)	26.95 (3.49)	67.24 (3.93)	5.81 (1.86)
	2022	16	0.31 (0.14)	28.75 (5.97)	71.25 (5.97)	0 (0)
strip	2020	48	1.29 (0.13)	68.83 (3.41)	18.13 (2.56)	13.04 (3.29)
	2021	50	0.84 (0.1)	48.28 (3.35)	24.24 (3.17)	27.48 (3.46)
	2022	20	1.17 (0.28)	47.57 (8.55)	50.61 (8.92)	1.82 (1.09)

Table S 7.5 Abundance of identified species for each of the sampling methods. Treatments included strip locations adjacent to crops with floral strips; control locations adjacent to crops without any enhancements; OSC locations adjacent to crops without any enhancements but were located on the same field as the strip; and semi-natural areas without management. OSC margins were at least 150 m away from any strips. Insects were collected in Manitoba Canada in the summers of 2020–2022.

[illegible]

Table S 7.5 cont'd

Family	Genus	Species	author	<u>blue vane</u>				<u>net</u>				<u>pan trap</u>				<u>pitfall trap</u>			
				control	natural	osc	strip	control	natural	osc	strip	control	natural	osc	strip	control	natural	osc	strip
Andrenidae	<i>Calliopsis</i>	<i>coloradensis</i>	Cresson 1878	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0
Andrenidae	<i>Perdita</i>	<i>albipennis</i>	Cresson 1868	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0
Andrenidae	<i>Perdita</i>	<i>bruneri</i>	Cockerell 1897	0	0	0	0	0	9	0	0	0	0	0	0	0	0	0	0
Andrenidae	<i>Perdita</i>	<i>swenki</i>	Crawford 1915	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0
Andrenidae	<i>Protandrena</i>			0	0	0	0	0	0	0	4	0	0	0	0	0	0	0	0
Andrenidae	<i>Protandrena</i>	<i>aestivalis</i>	(Provancher 1882)	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0
Andrenidae	<i>Protandrena</i>	<i>parvus</i>	(Robertson 1892)	0	0	1	3	0	1	1	0	0	0	1	1	0	0	0	0
Andrenidae	<i>Protandrena</i>	<i>perlaevis</i>	(Swenk and Cockerell 1907)	2	1	1	1	0	0	0	0	0	0	0	0	0	0	0	0
Andrenidae	<i>Protandrena</i>	<i>renimaculatus</i>	(Cockerell 1896)	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0
Apidae	<i>Anthophora</i>	<i>bomboides</i>	Kirby 1837	3	1	6	8	0	0	0	0	0	0	0	0	0	0	0	0
Apidae	<i>Anthophora</i>	<i>terminalis</i>	Cresson 1869	20	112	24	24	0	24	1	5	0	3	0	1	0	0	0	0
Apidae	<i>Anthophora</i>	<i>walshii</i>	Cresson 1869	2	1	0	3	0	0	0	0	0	0	0	0	0	0	0	0
Apidae	<i>Apis</i>	<i>mellifera</i>	Linnaeus 1758	319	220	584	1763	53	121	147	763	7	38	59	71	0	0	1	0
Apidae	<i>Bombus</i>			0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0
Apidae	<i>Bombus</i>	<i>bimaculatus</i>	Cresson 1863	15	19	24	18	0	2	0	5	1	0	0	1	0	0	0	0
Apidae	<i>Bombus</i>	<i>borealis</i>	Kirby 1837	323	306	541	761	5	41	4	32	3	7	7	15	0	0	1	0
Apidae	<i>Bombus</i>	<i>fervidus</i>	(Fabricius 1798)	6	1	4	11	0	0	0	0	0	1	0	1	0	0	0	0
Apidae	<i>Bombus</i>	<i>griseocollis</i>	(DeGeer 1773)	15	3	9	17	2	6	4	21	2	0	1	1	0	0	0	0
Apidae	<i>Bombus</i>	<i>huntii</i>	Greene 1860	1	1	3	2	0	1	1	5	0	0	0	0	0	0	0	0
Apidae	<i>Bombus</i>	<i>insularis</i>	(Smith 1861)	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0
Apidae	<i>Bombus</i>	<i>nevadensis</i>	Cresson 1874	5	3	2	17	1	0	0	1	0	0	0	0	1	0	0	0

Table S 7.5 cont'd

[illegible]

Table S 7.5 cont'd

Family	Genus	Species	author	blue vane				net				pan trap				pitfall trap			
				control	natural	osc	strip	control	natural	osc	strip	control	natural	osc	strip	control	natural	osc	strip
Apidae	<i>Melissodes</i>	<i>druriellus</i>	(Kirby 1802)	0	1	1	0	0	1	0	0	0	0	0	0	0	0	0	0
Apidae	<i>Melissodes</i>	<i>illatus</i>	Lovell and Cockerell 1906	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Apidae	<i>Melissodes</i>	<i>menuachus</i>	Cresson 1868	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Apidae	<i>Melissodes</i>	<i>rivalis</i>	Cresson 1872	3	2	2	2	0	4	0	0	0	0	0	0	0	0	0	0
Apidae	<i>Melissodes</i>	<i>snowii</i>	Cresson 1872	1	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0
Apidae	<i>Melissodes</i>	<i>trinodis</i>	Robertson 1901	179	197	269	344	2	6	1	1	5	1	1	1	0	1	0	0
Apidae	<i>Melissodes</i>	<i>wheeleri</i>	Cockerell 1906	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Apidae	<i>Nomada</i>	<i>composita</i>	Mitchell 1962	0	0	0	0	0	1	0	1	0	0	0	0	0	0	0	1
Apidae	<i>Nomada</i>	<i>crawfordi</i>	Cockerell 1905	0	0	1	1	1	0	0	0	0	0	0	0	0	0	0	0
Apidae	<i>Nomada</i>	<i>florilega</i>	Cockerell 1905	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0
Apidae	<i>Nomada</i>	<i>lehighensis</i>	Cockerell 1903	1	1	0	0	0	1	0	0	0	0	0	0	0	0	0	0
Apidae	<i>Nomada</i>	<i>quadrimaculata</i>	(Robertson 1903)	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0
Apidae	<i>Nomada</i>	<i>vicina</i>	Cresson 1863	0	1	0	0	0	1	0	0	0	0	1	0	0	0	0	0
Apidae	<i>Triepeolus</i>	<i>helianthi</i>	(Robertson 1897)	1	0	2	4	0	3	0	0	0	0	1	0	0	0	0	0
Apidae	<i>Triepeolus</i>	<i>obliteratus</i>	Graenicher 1911	1	1	0	1	0	1	0	0	0	0	0	0	0	0	0	0
Apidae	<i>Triepeolus</i>	<i>occidentalis</i>	(Cresson 1878)	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0
Carabidae				0	0	0	0	0	0	0	0	0	0	0	0	2	5	26	137
Carabidae	<i>Agonum</i>			0	0	0	0	0	0	0	0	0	0	0	0	179	45	305	594
Carabidae	<i>Amara</i>			0	0	0	0	0	0	0	0	0	0	0	0	176	15	838	735
Carabidae	<i>Anisodactylus</i>	<i>sanctaecrucis</i>	(Fabricius 1798)	0	0	0	0	0	0	0	0	0	0	0	0	1	0	19	62

Table S 7.5 cont'd

Family	Genus	Species	author	<u>blue vane</u>				<u>net</u>				<u>pan trap</u>				<u>pitfall trap</u>			
				control	natural	osc	strip	control	natural	osc	strip	control	natural	osc	strip	control	natural	osc	strip
Carabidae	<i>Bembidion</i>			0	0	0	0	0	0	0	0	0	0	0	0	3	0	12	28
Carabidae	<i>Bembidion</i>	<i>quadrimaculatum</i>	(Linnaeus 1761)	0	0	0	0	0	0	0	0	0	0	0	0	17	16	106	191
Carabidae	<i>Brachinus</i>			0	0	0	0	0	0	0	0	0	0	0	0	0	1	4	0
Carabidae	<i>Calosoma</i>			0	0	0	0	0	0	0	0	0	0	0	0	45	2	224	299
Carabidae	<i>Calosoma</i>	<i>calidum</i>	(Fabricius 1775)	0	0	0	0	0	0	0	0	0	0	0	0	17	2	35	118
Carabidae	<i>Carabus</i>	<i>granulatus</i>	Linnaeus 1758	0	0	0	0	0	0	0	0	0	0	0	0	17	150	15	4
Carabidae	<i>Chlaenius</i>			0	0	0	0	0	0	0	0	0	0	0	0	5	7	8	5
Carabidae	<i>Cymindis</i>			0	0	0	0	0	0	0	0	0	0	0	0	2	4	20	5
Carabidae	<i>Dicaelus</i>			0	0	0	0	0	0	0	0	0	0	0	0	28	2	18	1
Carabidae	<i>Diplocheila</i>			0	0	0	0	0	0	0	0	0	0	0	0	2	6	12	5
Carabidae	<i>Dyschirius</i>			0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1
Carabidae	<i>Elaphrus</i>			0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0
Carabidae	<i>Harpalus</i>			0	0	0	0	0	0	0	0	0	0	0	0	715	17	323	1521
Carabidae	<i>Loricera</i>			0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1
Carabidae	<i>Microlestes</i>			0	0	0	0	0	0	0	0	0	0	0	0	1	1	0	2
Carabidae	<i>Notiophilus</i>			0	0	0	0	0	0	0	0	0	0	0	0	0	0	2	0
Carabidae	<i>Patrobus</i>			0	0	0	0	0	0	0	0	0	0	0	0	0	2	0	0
Carabidae	<i>Pseudamara</i>	<i>arenaria</i>	(LeConte 1847)	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1
Carabidae	<i>Pterostichus</i>			0	0	0	0	0	0	0	0	0	0	0	0	946	49	5855	7254
Carabidae	<i>Pterostichus</i>	<i>melanarius</i>	(Illiger 1798)	0	0	0	0	0	0	0	0	0	0	0	0	865	327	1938	1381
Carabidae	<i>Stenolophus</i>			0	0	0	0	0	0	0	0	0	0	0	0	1	0	2	31

Table S 7.5 cont'd

Family	Genus	Species	author	<u>blue</u> <u>vane</u>	<u>net</u>	<u>pan</u> <u>trap</u>	<u>pitfall</u> <u>trap</u>	control	natural	osc	strip	control	natural	osc	strip	control	natural	osc	strip
Carabidae	<i>Synuchus</i>			0	0	0	0	0	0	0	0	0	0	0	0	0	1	1	0
Carabidae	<i>Synuchus</i>	<i>impunctatus</i>	(Say 1823)	0	0	0	0	0	0	0	0	0	0	0	0	0	8	0	0
Carabidae	<i>Tachys</i>			0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0
Colletidae	<i>Colletes</i>	<i>andrewsi</i>	Cockerell 1906	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Colletidae	<i>Colletes</i>	<i>brevicornis</i>	Robertson 1897	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0
Colletidae	<i>Colletes</i>	<i>hyalinus</i>	Provancher 1888	0	2	0	2	0	2	0	0	1	1	0	0	0	0	0	0
Colletidae	<i>Colletes</i>	<i>impunctatus</i>	Nylander 1852	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0
Colletidae	<i>Colletes</i>	<i>kincaidii</i>	Cockerell 1898	1	7	3	1	0	2	2	2	0	0	0	1	0	0	0	0
Colletidae	<i>Colletes</i>	<i>simulans</i>	Cockerell 1929	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0
Colletidae	<i>Hylaeus</i>			0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0
Colletidae	<i>Hylaeus</i>	<i>affinis</i>	(Smith 1853)	7	6	4	15	4	28	1	7	2	2	2	1	0	0	0	0
Colletidae	<i>Hylaeus</i>	<i>annulatus</i>	(Linnaeus 1758)	0	5	4	2	0	0	2	0	1	1	0	0	0	0	0	0
Colletidae	<i>Hylaeus</i>	<i>basalis</i>	(Smith 1853)	0	0	0	0	1	0	0	0	0	0	1	0	0	0	0	0
Colletidae	<i>Hylaeus</i>	<i>fedorica</i>	(Cockerell 1909)	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0
Colletidae	<i>Hylaeus</i>	<i>mesillae</i>	(Cockerell 1896)	0	1	0	0	0	12	0	0	0	1	0	0	0	0	0	0
Colletidae	<i>Hylaeus</i>	<i>mesillae/cressoni</i>		0	0	0	0	0	6	0	0	0	0	0	0	0	0	0	0
Colletidae	<i>Hylaeus</i>	<i>modestus</i>	Say 1837	0	1	2	1	0	2	0	1	0	2	1	0	0	0	0	0
Colletidae	<i>Hylaeus</i>	<i>verticillis</i>	(Cresson 1869)	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0
Halictidae	<i>Agapostemon</i>	<i>sericeus</i>	(Forster 1771)	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Halictidae	<i>Agapostemon</i>	<i>texasus</i>	Cresson 1872	52	20	11	15	0	3	0	2	2	1	21	21	0	0	0	0
Halictidae	<i>Augochlorella</i>	<i>aurata</i>	(Smith 1853)	0	0	0	0	0	0	0	0	0	0	1	1	0	0	0	0

Table S 7.5 cont'd

Family	Genus	Species	author	<u>blue</u>	<u>net</u>	<u>pan</u>	<u>pitfall</u>												
				vane control	natural	trap osc	trap strip	control	natural	osc	strip	control	natural	osc	strip	control	natural	osc	strip
Halictidae	<i>Dufourea</i>	<i>marginata</i>	(Cresson 1878)	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0
Halictidae	<i>Halictus</i>	<i>confusus</i>	Smith 1853	5	5	4	8	1	2	0	6	10	1	12	31	0	0	0	0
Halictidae	<i>Halictus</i>	<i>parallelus</i>	Say 1837	0	0	0	0	0	0	0	0	2	1	0	2	0	0	0	0
Halictidae	<i>Halictus</i>	<i>rubicundus</i>	Christ 1791	15	8	3	15	5	2	2	9	6	2	8	30	0	0	0	0
Halictidae	<i>Lasioglossum</i>	<i>absimile</i>	(Sandhouse 1924)	0	0	0	0	0	1	0	0	1	2	8	7	0	0	0	0
Halictidae	<i>Lasioglossum</i>	<i>admirandum</i>	(Sandhouse 1924)	0	0	0	1	0	0	0	0	0	0	0	1	0	0	0	0
Halictidae	<i>Lasioglossum</i>	<i>albipenne</i>	(Robertson 1890)	11	4	11	20	6	1	2	11	31	5	69	51	0	0	0	0
Halictidae	<i>Lasioglossum</i>	<i>athabascense</i>	Sandhouse 1933	0	1	0	0	0	0	0	0	0	2	0	0	0	0	0	0
Halictidae	<i>Lasioglossum</i>	<i>cinctipes</i>	Provancher 1888	0	2	1	0	0	0	0	1	3	2	3	7	0	0	0	0
Halictidae	<i>Lasioglossum</i>	<i>coriaceum</i>	Smith 1853	0	3	0	1	0	0	0	0	0	0	0	0	0	0	0	0
Halictidae	<i>Lasioglossum</i>	<i>ellisiae</i>	(Sandhouse 1924)	0	0	1	0	0	0	0	2	0	1	0	1	0	0	0	0
Halictidae	<i>Lasioglossum</i>	<i>ephialtum</i>	Gibbs 2010	0	0	0	3	0	0	0	0	0	0	1	0	0	0	0	0
Halictidae	<i>Lasioglossum</i>	<i>foxii</i>	(Robertson 1895)	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0
Halictidae	<i>Lasioglossum</i>	<i>laevissimum</i>	(Smith 1853)	0	0	0	1	0	0	0	0	0	0	1	1	0	0	0	0
Halictidae	<i>Lasioglossum</i>	<i>leucomus</i>	(Lovell 1908)	0	0	0	0	0	0	0	0	0	0	2	0	0	0	0	0
Halictidae	<i>Lasioglossum</i>	<i>leucozonium</i>	(Schrank 1781)	4	7	5	22	1	6	0	5	2	0	3	8	0	0	0	0
Halictidae	<i>Lasioglossum</i>	<i>lineatulum</i>	(Crawford 1906)	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0
Halictidae	<i>Lasioglossum</i>	<i>nigroviride</i>	(Graenicher 1911)	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Halictidae	<i>Lasioglossum</i>	<i>novascotiae</i>	Mitchell 1960	2	2	2	26	0	0	0	0	4	3	5	13	0	0	0	0
Halictidae	<i>Lasioglossum</i>	<i>oblongum</i>	(Lovell 1905)	0	1	0	0	0	0	0	0	0	0	1	0	0	0	0	0
Halictidae	<i>Lasioglossum</i>	<i>occidentale</i>	(Crawford 1902)	0	2	0	2	0	1	0	0	0	0	1	2	0	0	0	0

Table S 7.5 cont'd

Family	Genus	Species	author	blue vane				net				pan trap				pitfall trap			
				control	natural	osc	strip	control	natural	osc	strip	control	natural	osc	strip	control	natural	osc	strip
Halictidae	<i>Lasioglossum</i>	<i>paraforbesii</i>	McGinley 1986	30	12	11	29	0	1	0	3	3	0	18	41	0	0	0	0
Halictidae	<i>Lasioglossum</i>	<i>perpunctatum</i>	(Ellis 1913)	6	6	8	21	7	3	0	5	10	1	6	18	0	0	0	1
Halictidae	<i>Lasioglossum</i>	<i>pilosum</i>	(Smith 1853)	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0
Halictidae	<i>Lasioglossum</i>	<i>pruinsum</i>	(Robertson 1892)	2	0	1	3	0	0	0	0	1	0	4	4	0	0	0	0
Halictidae	<i>Lasioglossum</i>	<i>sagax</i>	(Sandhouse 1924)	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Halictidae	<i>Lasioglossum</i>	<i>semicaeruleum</i>	Cockerall 1895	0	0	2	6	0	0	0	6	38	1	31	92	0	0	1	0
Halictidae	<i>Lasioglossum</i>	<i>subviridatum</i>	(Cockerell 1938)	0	0	0	0	0	1	0	0	0	1	0	0	0	0	0	0
Halictidae	<i>Lasioglossum</i>	<i>succinipenne</i>	(Ellis 1913)	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0
Halictidae	<i>Lasioglossum</i>	<i>truncatum</i>	(Robertson 1901)	0	4	1	2	0	0	0	1	0	0	0	0	0	0	0	0
Halictidae	<i>Lasioglossum</i>	<i>zephyrus</i>	(Smith 1853)	0	18	0	0	0	2	1	0	0	23	1	0	0	0	0	0
Halictidae	<i>Lasioglossum</i>	<i>zonulus</i>	(Smith 1848)	11	26	41	40	2	6	0	1	4	5	10	11	0	0	0	0
Halictidae	<i>Sphecodes</i>	<i>illinoensis</i>	(Robertson 1903)	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Megachilidae	<i>Anthidium</i>	<i>clypeodentatum</i>	Swenk 1914	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0
Megachilidae	<i>Anthidium</i>	<i>tenuiflorae</i>	Cockerell 1907	0	0	1	1	0	0	0	0	0	0	0	0	0	0	0	0
Megachilidae	<i>Coelioxys</i>	<i>funerarius</i>	Smith 1854	1	2	0	0	1	1	0	1	0	0	0	0	0	0	0	0
Megachilidae	<i>Coelioxys</i>	<i>modestus</i>	Smith 1854	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0
Megachilidae	<i>Coelioxys</i>	<i>rufitarsis</i>	Smith 1854	0	0	0	0	0	1	0	2	0	1	1	0	0	0	0	0
Megachilidae	<i>Dianthidium</i>	<i>pudicum</i>	(Cresson 1879)	1	0	1	0	0	0	0	0	0	0	0	1	0	0	0	0
Megachilidae	<i>Heriades</i>	<i>carinata</i>	Cresson 1864	4	15	22	12	1	26	12	23	1	7	8	6	0	0	0	0
Megachilidae	<i>Heriades</i>	<i>variolora</i>	(Cresson 1872)	0	0	0	0	0	8	0	0	0	0	0	1	0	0	0	0
Megachilidae	<i>Hoplitis</i>	<i>pilosifrons</i>	(Cresson 1864)	8	4	21	28	1	1	0	3	1	0	4	7	0	0	0	0

Table S 7.5 cont'd

Family	Genus	Species	author	blue vane				net				pan trap				pitfall trap			
				control	natural	osc	strip	control	natural	osc	strip	control	natural	osc	strip	control	natural	osc	strip
Megachilidae	<i>Hoplitis</i>	<i>producta</i>	(Cresson 1864)	2	1	1	2	0	2	0	2	2	2	3	0	0	0	0	0
Megachilidae	<i>Hoplitis</i>	<i>spoliata</i>	(Provancher 1888)	0	0	0	0	0	1	0	0	0	0	1	0	0	0	0	0
Megachilidae	<i>Megachile</i>	<i>brevis</i>	Say 1837	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0
Megachilidae	<i>Megachile</i>	<i>campanulae</i>	(Robertson 1903)	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Megachilidae	<i>Megachile</i>	<i>centuncularis</i>	(Linnaeus 1758)	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Megachilidae	<i>Megachile</i>	<i>frigida</i>	Smith 1853	0	0	0	0	1	3	0	4	0	0	0	0	0	0	0	0
Megachilidae	<i>Megachile</i>	<i>inermis</i>	Provancher 1888	4	21	10	12	0	12	5	8	1	7	2	1	0	0	0	0
Megachilidae	<i>Megachile</i>	<i>latimanus</i>	Say 1823	9	2	11	14	1	2	7	9	2	1	7	5	0	0	0	1
Megachilidae	<i>Megachile</i>	<i>melanophaea</i>	Smith 1853	1	1	2	2	0	2	0	1	1	0	0	0	0	0	0	0
Megachilidae	<i>Megachile</i>	<i>mendica</i>	Cresson 1878	0	0	0	0	0	0	0	2	0	0	0	0	0	0	0	0
Megachilidae	<i>Megachile</i>	<i>montivaga</i>	Cresson 1878	0	1	0	1	0	0	0	0	0	0	0	0	0	0	0	0
Megachilidae	<i>Megachile</i>	<i>perihirta</i>	Cockerell 1898	1	1	4	3	1	0	0	1	0	0	2	0	0	0	0	0
Megachilidae	<i>Megachile</i>	<i>pugnata</i>	Say 1837	1	0	1	2	0	0	0	1	0	0	0	0	0	0	0	0
Megachilidae	<i>Megachile</i>	<i>relativa</i>	Cresson 1878	0	1	0	1	0	8	6	5	1	0	0	2	0	0	0	0
Megachilidae	<i>Megachile</i>	<i>rotundata</i>	(Fabricius 1787)	0	0	2	0	1	1	0	4	0	0	5	2	0	0	0	0
Megachilidae	<i>Megachile</i>	<i>texana</i>	Cresson 1878	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0
Megachilidae	<i>Osmia</i>	<i>atriventris</i>	Cresson 1864	0	3	0	1	0	0	0	0	0	0	0	0	0	0	0	0
Megachilidae	<i>Osmia</i>	<i>buchephalae</i>	Cresson 1864	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0
Megachilidae	<i>Osmia</i>	<i>simillima</i>	Smith 1853	3	8	3	3	0	1	0	0	0	0	0	0	0	0	0	0
Megachilidae	<i>Osmia</i>	<i>tersula</i>	Cockerell 1912	1	2	0	0	0	0	0	0	0	0	1	0	0	0	0	0
Megachilidae	<i>Stelis</i>	<i>coarctatus</i>	Crawford 1916	0	0	0	0	0	0	0	0	1	1	0	0	0	0	0	0

Table S 7.5 cont'd

Family	Genus	Species	author	blue vane				net				pan trap				pitfall trap			
				control	natural	osc	strip	control	natural	osc	strip	control	natural	osc	strip	control	natural	osc	strip
Megachilidae	<i>Stelis</i>	<i>lateralis</i>	Cresson 1864	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Melittidae	<i>Macropis</i>	<i>nuda</i>	(Provancher 1882)	0	0	0	0	10	0	0	7	0	0	0	0	0	0	0	0
Syrphidae	<i>Allograpta</i>	<i>obliqua</i>	(Say 1823)	1	1	0	0	0	0	0	2	0	0	0	0	0	0	0	0
Syrphidae	<i>Eristalis</i>			0	0	1	1	0	0	0	0	0	0	0	0	0	0	0	0
Syrphidae	<i>Eristalis</i>	<i>dimidiata</i>	Wiedemann 1830	0	0	0	0	29	7	29	44	0	0	0	0	0	0	0	0
Syrphidae	<i>Eristalis</i>	<i>stipator</i>	Osten Sacken 1877	3	4	3	6	18	4	7	19	0	0	3	5	0	0	0	0
Syrphidae	<i>Eristalis</i>	<i>tenax</i>	(Linnaeus 1758)	1	0	1	1	5	2	0	14	0	0	0	0	0	0	0	0
Syrphidae	<i>Eupeodes</i>	<i>americanus</i>	(Wiedemann 1830)	1	4	1	1	0	1	2	7	2	2	0	0	1	1	0	1
Syrphidae	<i>Eupeodes</i>	<i>latifasciatus</i>	(Macquart 1829)	35	107	84	51	0	3	2	6	4	1	3	6	0	0	0	0
Syrphidae	<i>Eupeodes</i>	<i>volucris</i>	Osten Sacken 1877	14	13	14	11	1	1	0	2	4	2	5	9	0	0	0	0
Syrphidae	<i>Eurimyia</i>	<i>stipata</i>	(Walker 1849)	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Syrphidae	<i>Helophilus</i>			0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0
Syrphidae	<i>Helophilus</i>	<i>fasciatus</i>	(Walker 1849)	1	4	5	3	1	3	3	4	1	0	0	2	0	0	0	0
Syrphidae	<i>Helophilus</i>	<i>hybridus</i>	Loew 1846	0	0	0	0	1	2	1	4	0	0	0	0	0	0	0	0
Syrphidae	<i>Helophilus</i>	<i>latifrons</i>	(Loew 1863)	56	23	36	38	8	1	3	22	4	0	16	13	0	0	0	0
Syrphidae	<i>Helophilus</i>	<i>obscurus</i>	(Loew 1863)	8	10	21	29	0	1	0	1	2	0	1	2	0	0	0	0
Syrphidae	<i>Paragus</i>			0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0
Syrphidae	<i>Parhelophilus</i>			1	6	0	0	0	0	0	0	0	1	0	0	0	0	0	0
Syrphidae	<i>Platycheirus</i>			1	3	2	1	0	1	1	1	1	1	1	0	0	0	0	0
Syrphidae	<i>Rhingia</i>	<i>nasica</i>	Say 1823	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Syrphidae	<i>Sericomyia</i>	<i>militarsis</i>	Walker 1849	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0

Table S 7.5 cont'd

Family	Genus	Species	author	<u>blue vane</u>				<u>net</u>				<u>pan trap</u>				<u>pitfall trap</u>			
				control	natural	osc	strip	control	natural	osc	strip	control	natural	osc	strip	control	natural	osc	strip
Syrphidae	<i>Sphaerophoria</i>	<i>contigua</i>	Macquart 1847	2	1	7	7	6	2	2	8	1	0	3	6	0	0	0	0
Syrphidae	<i>Sphaerophoria</i>	<i>philanthus</i>	(Meigen 1822)	3	3	6	9	8	0	3	6	1	2	1	0	0	0	0	0
Syrphidae	<i>Syrpitta</i>	<i>pipiens</i>	(Linnaeus 1758)	0	3	0	6	3	4	9	7	0	1	3	2	0	0	0	0
Syrphidae	<i>Syrphus</i>	<i>ribesii</i>	(Linnaeus 1758)	6	9	5	1	0	1	0	2	0	0	2	0	0	0	0	0
Syrphidae	<i>Toxomerus</i>	<i>geminatus</i>	Say 1823	49	93	17	12	7	27	4	2	5	5	47	41	0	0	0	0
Syrphidae	<i>Toxomerus</i>	<i>marginatus</i>	Say 1823	83	198	193	194	62	80	27	192	120	43	265	294	3	1	6	1
Syrphidae	<i>Tropidia</i>	<i>quadrata</i>	(Say 1824)	0	0	0	0	0	0	0	3	0	0	0	0	0	0	0	0
Syrphidae	<i>Xylota</i>	<i>subfasciata</i>	Loew 1866	1	4	1	2	0	0	0	0	0	0	0	1	0	0	0	0

Table S 7.6 Statistical results of a post hoc test of a permutational analysis of variance on bee and Syrphidae communities collected at sites of three treatments during the summers of 2020–2022. Data were collected in Manitoba Canada. Treatments included strip locations adjacent to crops with floral strips; control locations adjacent to crops without any enhancements; and semi-natural areas without management. Collection methods included net insects collected along a 50 metre transect for 10 minutes of active collection with a net; blue vane insects that were collected using blue vane traps that were left in fields for 7 days; and pan traps insects that were collected with pan traps of three colours (blue white and yellow) and were installed at sites during the net samples

Taxon	pairwise comparison		df	F	<i>p</i> value	P adjusted
Bees	control_2020	control_2021	1	6.032	0.001	0.004
	control_2020	control_2022	1	1.326	0.167	0.273
	control_2020	natural_2020	1	4.171	0.004	0.013
	control_2020	natural_2021	1	10.097	0.001	0.004
	control_2020	natural_2022	1	1.418	0.166	0.273
	control_2020	strip_2020	1	29.425	0.001	0.004
	control_2020	strip_2021	1	28.245	0.001	0.004
	control_2020	strip_2022	1	3.845	0.005	0.015
	control_2021	control_2022	1	0.347	0.982	0.982
	control_2021	natural_2020	1	1.460	0.144	0.268
	control_2021	natural_2021	1	2.579	0.018	0.050
	control_2021	natural_2022	1	1.252	0.190	0.297
	control_2021	strip_2020	1	15.436	0.001	0.004
	control_2021	strip_2021	1	14.796	0.001	0.004
	control_2021	strip_2022	1	1.588	0.109	0.231
	control_2022	natural_2020	1	0.624	0.659	0.719
	control_2022	natural_2021	1	0.550	0.834	0.858
	control_2022	natural_2022	1	0.883	0.548	0.617
	control_2022	strip_2020	1	1.237	0.238	0.317
	control_2022	strip_2021	1	1.150	0.256	0.318
	control_2022	strip_2022	1	1.012	0.372	0.432
	natural_2020	natural_2021	1	2.618	0.027	0.069
	natural_2020	natural_2022	1	1.125	0.209	0.301
	natural_2020	strip_2020	1	14.745	0.001	0.004
	natural_2020	strip_2021	1	13.908	0.001	0.004
	natural_2020	strip_2022	1	2.003	0.069	0.166
	natural_2021	natural_2022	1	1.024	0.291	0.349
	natural_2021	strip_2020	1	8.202	0.001	0.004

Table S 7.6 cont'd

Taxon	pairwise comparison		df	F	<i>p</i> value	P adjusted
Bees	natural_2021	strip_2021	1	7.933	0.001	0.004
	natural_2021	strip_2022	1	1.266	0.203	0.301
	natural_2022	strip_2020	1	1.812	0.105	0.231
	natural_2022	strip_2021	1	1.748	0.134	0.268
	natural_2022	strip_2022	1	1.512	0.149	0.268
	strip_2020	strip_2021	1	0.510	0.798	0.845
	strip_2020	strip_2022	1	1.311	0.228	0.316
	strip_2021	strip_2022	1	1.228	0.248	0.318
Syrphids	control_2020	control_2021	1	6.286	0.001	0.009
	control_2020	control_2022	1	2.465	0.062	0.172
	control_2020	natural_2020	1	1.460	0.180	0.381
	control_2020	natural_2021	1	2.274	0.057	0.171
	control_2020	natural_2022	1	0.339	0.852	0.929
	control_2020	strip_2020	1	2.192	0.072	0.175
	control_2020	strip_2021	1	1.436	0.220	0.440
	control_2020	strip_2022	1	0.428	0.750	0.900
	control_2021	control_2022	1	0.751	0.457	0.782
	control_2021	natural_2020	1	7.501	0.001	0.009
	control_2021	natural_2021	1	4.832	0.005	0.036
	control_2021	natural_2022	1	0.254	0.929	0.954
	control_2021	strip_2020	1	10.004	0.001	0.009
	control_2021	strip_2021	1	10.585	0.001	0.009
	control_2021	strip_2022	1	0.773	0.454	0.782
	control_2022	natural_2020	1	2.931	0.044	0.171
	control_2022	natural_2021	1	2.317	0.073	0.175
	control_2022	natural_2022	1	0.703	0.571	0.824
	control_2022	strip_2020	1	2.789	0.053	0.171
	control_2022	strip_2021	1	2.962	0.046	0.171
	control_2022	strip_2022	1	0.914	0.416	0.782
	natural_2020	natural_2021	1	1.526	0.176	0.381
	natural_2020	natural_2022	1	0.382	0.807	0.908
	natural_2020	strip_2020	1	2.480	0.057	0.171
	natural_2020	strip_2021	1	0.763	0.507	0.794
	natural_2020	strip_2022	1	0.541	0.659	0.840
	natural_2021	natural_2022	1	0.167	0.954	0.954

	natural_2021	strip_2020	1	4.072	0.007	0.042
Table S 7.6 cont'd						
Taxon	pairwise comparison		df	F	<i>p</i> value	P adjusted
Syrphids	natural_2021	strip_2021	1	2.632	0.040	0.171
	natural_2021	strip_2022	1	0.789	0.478	0.782
	natural_2022	strip_2020	1	0.558	0.639	0.840
	natural_2022	strip_2021	1	0.492	0.677	0.840
	natural_2022	strip_2022	1	0.498	0.663	0.840
	strip_2020	strip_2021	1	0.691	0.572	0.824
	strip_2020	strip_2022	1	0.245	0.914	0.954
	strip_2021	strip_2022	1	0.374	0.800	0.908

Table S 7.7 Species composition of the seed mix planted in the enhanced hay fields. Seed mix was designed by the Xerces Society of North America and was mixed by Northstar seeds in Manitoba Canada. Seed mix was planted at sites in the summer of 2019. 10 – 5 alfalfa blend includes five varieties of alfalfa including: 30% robust 30% haygrazer 15% response WT 15% rugged ST and 10% sidewinder. Robust: multi-leaf variety of alfalfa that is selected for high winter survivability and high yields and quality. Rugged ST: salt tolerant variety that has increased germination rates in saline conditions. Response WT: high yield variety with branched roots that can handle higher moisture levels than conventional varieties. Haygrazer: variety contains a sunken crown and fibrous roots that gives this variety a higher tolerance to high traffic conditions. Sidewinder: creeping root variety with high yields and tolerant of drought conditions.

common name	species	Percent of mix
10 – 5 blend Alfalfa	<i>Medicago sativa</i>	45
Rugged ST Alfalfa	<i>Medicago sativa</i>	18
Fleet meadow brome grass	<i>Bromopsis biebersteinii</i>	10
intermediate wheatgrass	<i>Thinopyrum intermedium</i>	10
Cicer Milkvetch	<i>Astragalus cicer</i>	8
slender wheatgrass	<i>Elymus trachycaulus</i>	6
Alsike clover	<i>Trifolium hybridum</i>	1
single cut red clover	<i>Trifolium pratense</i>	1
double cut red clover	<i>Trifolium pratense</i>	1

Table S 7.8 Abundance of each species identified from sites of three treatments. Bees were collected in Western Manitoba from 2020 – 2022. Treatments included graze; locations that are rotationally grazed by cattle natural; semi-natural sites that have not been managed for at least 5 years and hay; fields that have been planted with an enhanced hay mixture. Species trait data are largely extracted from Gibbs et al. (2023) a checklist of the bees (Hymenoptera: Apoidea) of Manitoba. Transect type included random: randomly placed transects within the field and selective: transects were placed purposefully in areas with high floral abundance bees visible or in areas where the observer thought that capturing bees would be likely.

Family	Genus	Species	author	sociality	nesting	lecty	graze	hay	natural
Andrenidae	<i>Andrena</i>	<i>chromotricha</i>	Cockerell 1899	solitary	ground	oligolectic	2	1	0
Andrenidae	<i>Andrena</i>	<i>campanulae</i>	Viereck and Cockerell 1914	solitary	ground	oligolectic	0	0	1
Andrenidae	<i>Andrena</i>	<i>lupinorum</i>	Cockerell 1906	solitary	ground	oligolectic	0	0	4
Andrenidae	<i>Andrena</i>	<i>nubecula</i>	Cockerell 1931	solitary	ground	oligolectic	0	0	1
Andrenidae	<i>Andrena</i>	<i>regularis</i>	Malloch 1917	solitary	ground	polylectic	1	0	0
Andrenidae	<i>Andrena</i>	<i>runcinatae</i>	Cockerell 1906	solitary	ground	polylectic	3	0	5
Andrenidae	<i>Andrena</i>	<i>thaspii</i>	Graenicher 1903	solitary	ground	polylectic	2	0	2
Andrenidae	<i>Andrena</i>	<i>wilkella</i>	(Kirby 1802)	solitary	ground	oligolectic	0	0	1
Andrenidae	<i>Calliopsis</i>	<i>coloradensis</i>	Cresson 1878	solitary	ground	oligolectic	7	0	1
Andrenidae	<i>Perdita</i>	<i>bruneri</i>	Cockerell 1897	solitary	ground	oligolectic	76	0	9
Andrenidae	<i>Perdita</i>	<i>swenki</i>	Crawford 1915	solitary	ground	oligolectic	31	0	2
Andrenidae	<i>Protandrena</i>	<i>aestivalis</i>	(Provancher 1882)	solitary	ground	oligolectic	4	0	2
Andrenidae	<i>Protandrena</i>	<i>parva</i>	(Robertson 1892)	solitary	ground	polylectic	2	0	0
Andrenidae	<i>Protandrena</i>	<i>renimaculata</i>	(Cockerell 1896)	solitary	ground	oligolectic	1	0	0
Andrenidae	<i>Protandrena</i>						1	0	0
total Andrenidae							130	1	28

Table S 7.8 cont'd

Family	Genus	Species	author	sociality	nesting	lecty	graze	hay	natural
Apidae	<i>Anthophora</i>	<i>terminalis</i>	Cresson 1869	solitary	wood	polylectic	0	0	3
Apidae	<i>Apis</i>	<i>mellifera</i>	Linnaeus 1758	eusocial	hive	polylectic	17	32	20
Apidae	<i>Bombus</i>	<i>borealis</i>	Kirby 1837	eusocial	hive	polylectic	14	0	23
Apidae	<i>Bombus</i>	<i>griseocollis</i>	(DeGeer 1773)	eusocial	hive	polylectic	7	0	4
Apidae	<i>Bombus</i>	<i>huntii</i>	Greene 1860	eusocial	hive	polylectic	0	0	1
Apidae	<i>Bombus</i>	<i>insularis</i>	(Smith 1861)		hive		0	0	9
Apidae	<i>Bombus</i>	<i>rufocinctus</i>	Cresson 1863	eusocial	hive	polylectic	9	1	32
Apidae	<i>Bombus</i>	<i>ternarius</i>	Say 1837	eusocial	hive	polylectic	29	1	32
Apidae	<i>Bombus</i>	<i>terricola</i>	Kirby 1837	eusocial	hive	polylectic	7	3	15
Apidae	<i>Bombus</i>	<i>vagans</i>	Smith 1854	eusocial	hive	polylectic	6	1	1
Apidae	<i>Ceratina</i>	<i>calcarata</i>	Robertson 1900	subsocial	stem	polylectic	0	0	1
Apidae	<i>Epeolus</i>	<i>minimus</i>	(Robertson 1902)		ground		0	0	4
Apidae	<i>Melissodes</i>	<i>agilis</i>	Cresson 1878	solitary	ground	oligolectic	2	0	2
Apidae	<i>Melissodes</i>	<i>druriellus</i>	(Kirby 1802)	solitary	ground	oligolectic	1	0	1
Apidae	<i>Melissodes</i>	<i>menuachus</i>	Cresson 1868	solitary	ground	oligolectic	1	0	0
Apidae	<i>Melissodes</i>	<i>rivalis</i>	Cresson 1872	solitary	ground	oligolectic	7	0	4
Apidae	<i>Triepeolus</i>	<i>obliteratus</i>	Graenicher 1911		ground		1	0	0
Apidae	<i>Triepeolus</i>	<i>occidentalis</i>	(Cresson 1878)		ground		0	0	1
total Apidae							101	38	153

Table S 7.8 cont'd

Family	Genus	Species	author	sociality	nesting	lecty	graze	hay	natural
Colletidae	<i>Colletes</i>	<i>andrewsi</i>	Cockerell 1906	solitary	ground	oligolectic	1	0	0
Colletidae	<i>Colletes</i>	<i>hyalinus</i>	Provancher 1888	solitary	ground	polylectic	2	0	2
Colletidae	<i>Colletes</i>	<i>kincaidii</i>	Cockerell 1898	solitary	ground	polylectic	0	0	1
Colletidae	<i>Colletes</i>	<i>simulans</i>	Cockerell 1929	solitary	ground	oligolectic	0	0	1
Colletidae	<i>Hylaeus</i>	<i>affinis</i>	(Smith 1853)	solitary	stem	polylectic	0	0	1
Colletidae	<i>Hylaeus</i>	<i>mesillae</i>	(Cockerell 1896)	solitary	stem	polylectic	0	0	8
Colletidae	<i>Hylaeus</i>	<i>verticalis</i>	(Cresson 1869)	solitary	stem	polylectic	0	0	1
total Colletidae							3	0	14
Halictidae	<i>Agapostemon</i>	<i>sericeus</i>	(Forster 1771)	solitary	ground	polylectic	1	0	0
Halictidae	<i>Agapostemon</i>	<i>subtilior</i>	Cockerell 1898	solitary	ground	polylectic	1	0	3
Halictidae	<i>Dufourea</i>	<i>marginata</i>	(Cresson 1878)	solitary	ground	oligolectic	0	0	1
Halictidae	<i>Halictus</i>	<i>confusus</i>	Smith 1853	eusocial	ground	polylectic	3	0	1
Halictidae	<i>Halictus</i>	<i>rubicundus</i>	Christ 1791	eusocial	ground	polylectic	4	0	2
Halictidae	<i>Lasioglossum</i>	<i>albipenne</i>	(Robertson 1890)	eusocial	ground	polylectic	0	0	1
Halictidae	<i>Lasioglossum</i>	<i>leucocomus</i>	(Lovell 1908)	eusocial	ground	polylectic	2	0	0
Halictidae	<i>Lasioglossum</i>	<i>leucozonium</i>	(Schrank 1781)	solitary	ground	polylectic	1	0	4
Halictidae	<i>Lasioglossum</i>	<i>paraforbesii</i>	McGinley 1986	solitary	ground	polylectic	0	0	1
Halictidae	<i>Lasioglossum</i>	<i>perpunctatum</i>	(Ellis 1913)	eusocial	ground	polylectic	1	0	3
Halictidae	<i>Lasioglossum</i>	<i>pilosum</i>	(Smith 1853)	eusocial	ground	polylectic	1	0	0
Halictidae	<i>Lasioglossum</i>	<i>succinipenne</i>	(Ellis 1913)	eusocial	ground	polylectic	1	0	0
total Halictidae							15	0	16

Table S 7.8 cont'd

Family	Genus	Species	author	sociality	nesting	lecty	graze	hay	natural
Megachilidae	<i>Anthidium</i>	<i>clypeodentatum</i>	Swenk 1914	solitary	ground	polylectic	0	0	1
Megachilidae	<i>Coelioxys</i>	<i>rufitarsis</i>	Smith 1854		ground		0	0	1
Megachilidae	<i>Heriades</i>	<i>carinata</i>	Cresson 1864	solitary	stem	polylectic	4	0	11
Megachilidae	<i>Heriades</i>	<i>variolosa</i>	(Cresson 1872)	solitary	stem	oligolectic	1	0	8
Megachilidae	<i>Hoplitis</i>	<i>pilosifrons</i>	(Cresson 1864)	solitary	stem	polylectic	0	0	1
Megachilidae	<i>Hoplitis</i>	<i>producta</i>	(Cresson 1864)	solitary	stem	polylectic	1	0	1
Megachilidae	<i>Hoplitis</i>	<i>spoliata</i>	(Provancher 1888)	solitary	stem	polylectic	0	0	1
Megachilidae	<i>Megachile</i>	<i>brevis</i>	Say 1837	solitary	cavity	polylectic	0	0	2
Megachilidae	<i>Megachile</i>	<i>frigida</i>	Smith 1853	solitary	cavity	polylectic	0	0	1
Megachilidae	<i>Megachile</i>	<i>inermis</i>	Provancher 1888	solitary	cavity	polylectic	9	0	11
Megachilidae	<i>Megachile</i>	<i>latimanus</i>	Say 1823	solitary	ground	polylectic	2	1	0
Megachilidae	<i>Megachile</i>	<i>melanophaea</i>	Smith 1853	solitary	ground	polylectic	1	0	0
Megachilidae	<i>Megachile</i>	<i>montivaga</i>	Cresson 1878	solitary	stem	polylectic	1	0	0
Megachilidae	<i>Megachile</i>	<i>relativa</i>	Cresson 1878	solitary	cavity	polylectic	2	0	3
Megachilidae	<i>Megachile</i>	<i>rotundata</i>	(Fabricius 1787)	solitary	cavity	polylectic	0	0	1
Megachilidae	<i>Osmia</i>	<i>simillima</i>	Smith 1853	solitary	cavity	polylectic	0	0	1
total Megachilidae							21	1	43

SUPPLEMENTARY FIGURES

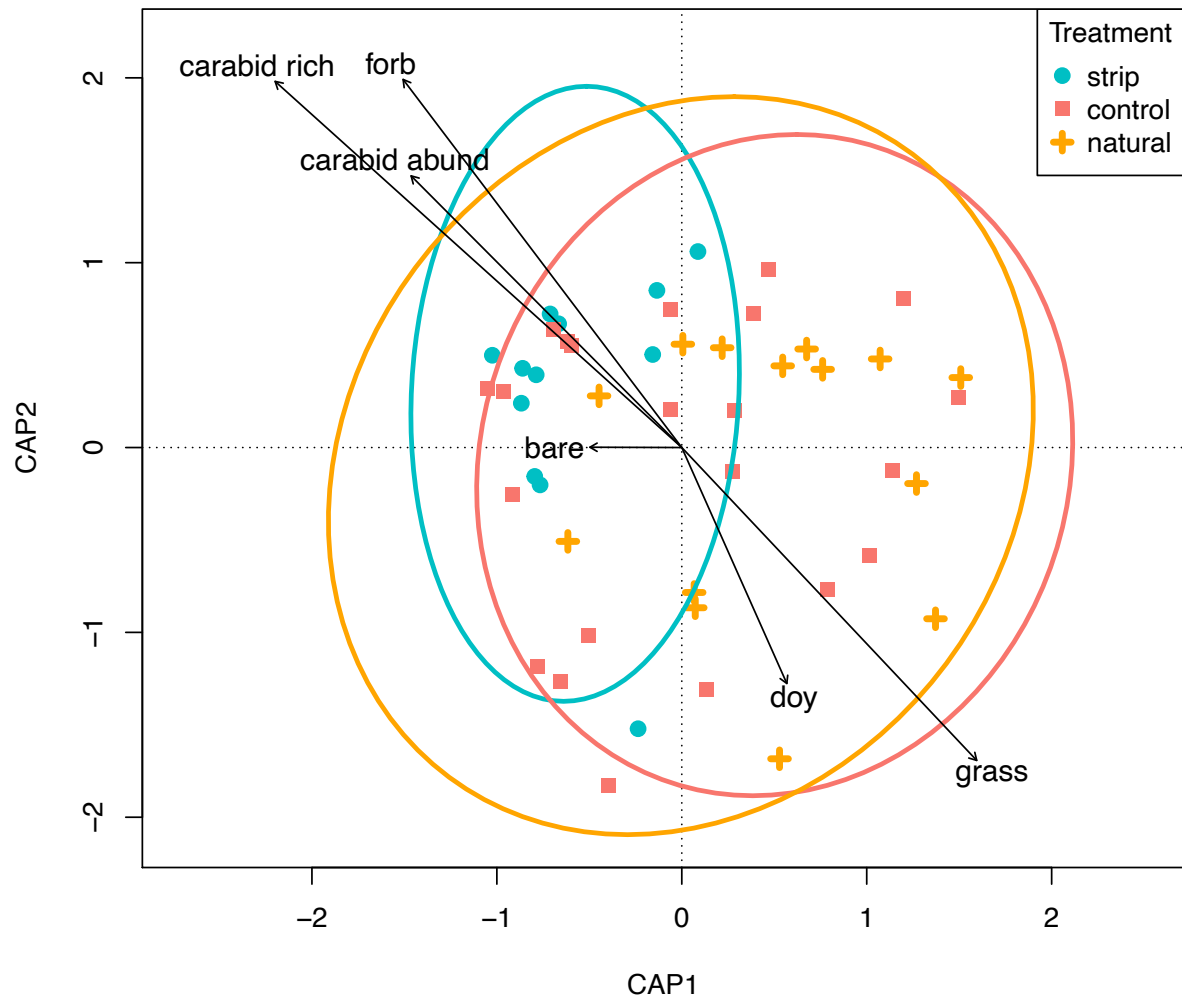


Figure S 7.1 Distance-based redundancy analysis with Bray-Curtis dissimilarity showing the abundance of ground beetle species at sites from three treatments. Treatments included strip locations adjacent to crops with floral strip; control locations adjacent to crops without any enhancements; and natural semi-natural areas without management. Ellipses represent 95% confidence intervals for each treatment. Vector labels: forb percent ground cover of forbs found in the vegetation transects; carabid_rich number of Carabidae species found in the pitfall samples; carabid_abund number of Carabidae specimens found in the pitfall samples; bare percentage of bare ground cover found in the vegetation transects; doy numerical day of the year when the pitfall sample was collected; grass percent ground cover of grasses found in the vegetation transects. Sample sizes: strip $n=12$; control $n=22$; natural $n=14$. Figure was created using the ordiplot function from the vegan package in R (Oksanen et al. 2019; R Core Team 2023).

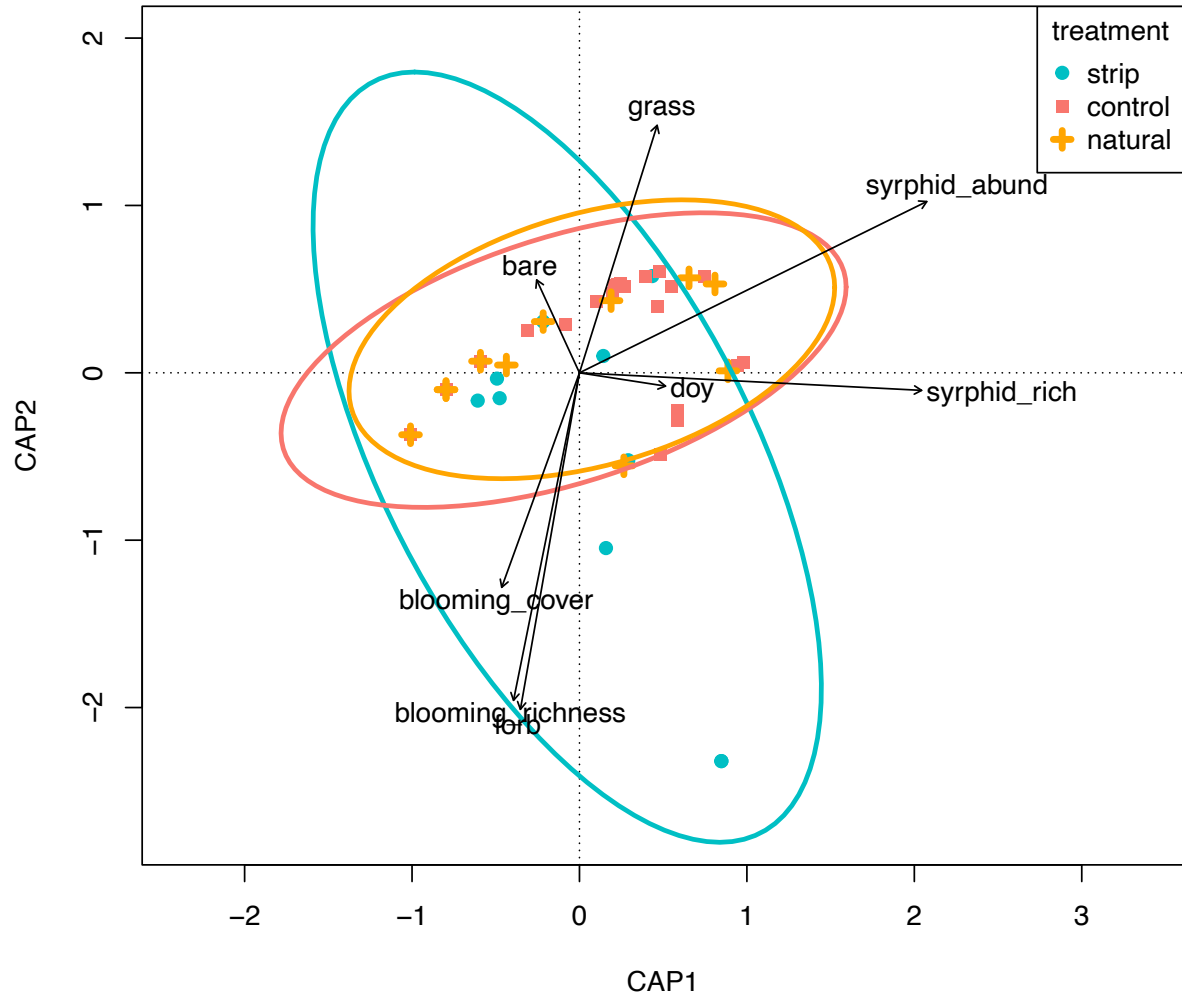


Figure S 7.2 Distance-based redundancy analysis with Bray-Curtis dissimilarity showing the abundance of syrphid species at sites from three treatments. Treatments included strip locations adjacent to crops with floral strip; control locations adjacent to crops without any enhancements; and natural semi-natural areas without management. Ellipses represent 95% confidence intervals for each treatment. Vector labels: grass percent ground cover of grasses found in the vegetation transects; syrphid_abund number of Syrphidae specimens found in the pan trap samples; syrphid_rich number of Syrphidae species found in the pan trap samples; doy numerical day of the year when the pan trap sample was collected; blooming_cover percent ground cover of blooming forbs found in the vegetation transects; blooming_richness number of blooming forb species found in the vegetation transects; forb percent ground cover of forbs found in the vegetation transects; bare percent bare ground cover found in the vegetation transects. Sample sizes: strip $n=12$; control $n=26$; natural $n=11$. Figure was created using the ordiplot function from the vegan package in R (Oksanen et al. 2019; R Core Team 2023).

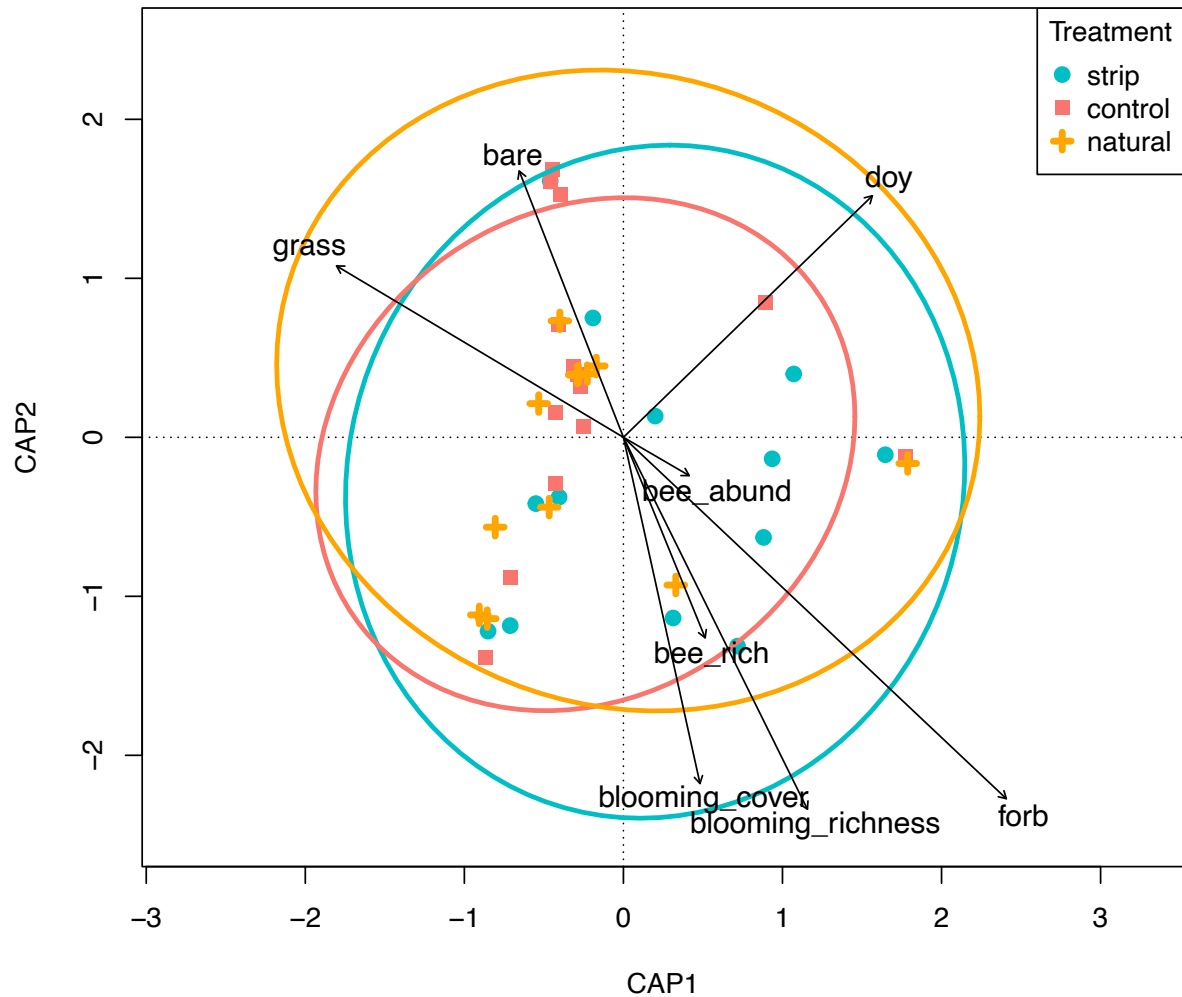


Figure S 7.3 Distance-based redundancy analysis with Bray-Curtis dissimilarity showing the abundance of bee species collected with pan traps from sites from three treatments. Ellipses represent 95% confidence intervals for each treatment. Vector labels: day numerical day of the year when the pan trap sample was collected; bee_abund number of bee specimens found in the pan trap samples; bee_rich number of bee species found in the pan trap samples; forb percent ground cover of forbs found in the vegetation transects; blooming_cover percent ground cover of blooming forbs found in the vegetation transects; blooming_richness number of blooming forb species found in the vegetation transects; grass percent ground cover of grasses found in the vegetation transects; bare percentage of bare ground cover found in the vegetation transects. Sample sizes: strip $n=13$; control $n=17$; natural $n=13$. Figure was created using the ordiplot function from the vegan package in R (Oksanen et al. 2019; R Core Team 2023).

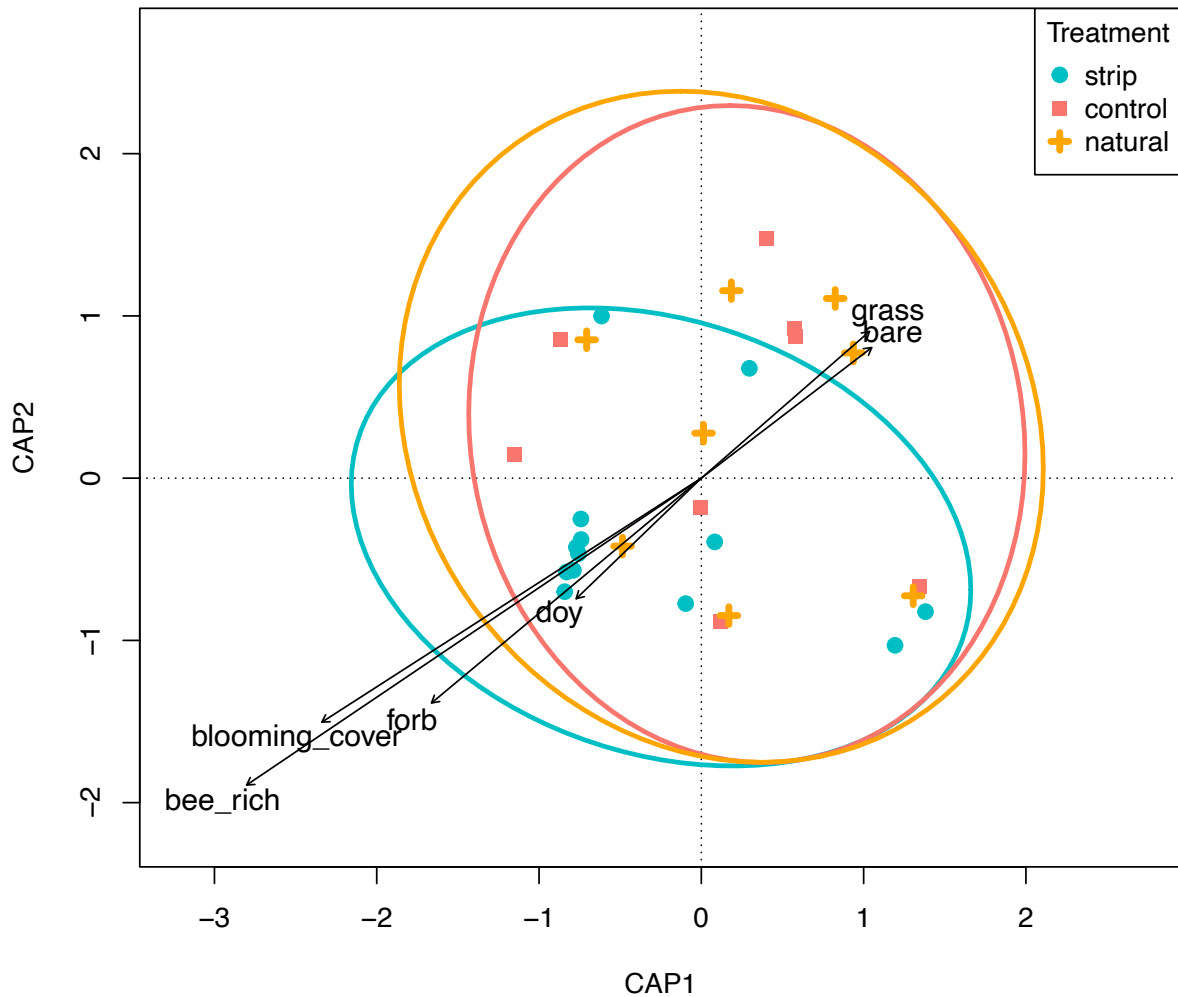


Figure S 7.4 Distance-based redundancy analysis with Bray-Curtis dissimilarity showing the abundance of bee species collected with nets from sites from three treatments. Ellipses represent 95% confidence intervals for each treatment. Vector labels: grass percent ground cover of grasses found in the vegetation transects; bare percentage of bare ground cover found in the vegetation transects; doy numerical day of the year when the net sample was collected; forb percent ground cover of forbs found in the vegetation transects; blooming_cover percent ground cover of blooming forbs found in the vegetation transects; bee_rich number of bee species found in the net samples. Sample sizes: strip $n=13$; control $n=8$; natural $n=8$. Figure was created using the ordiplot function from the vegan package in R (Oksanen et al. 2019; R Core Team 2023).

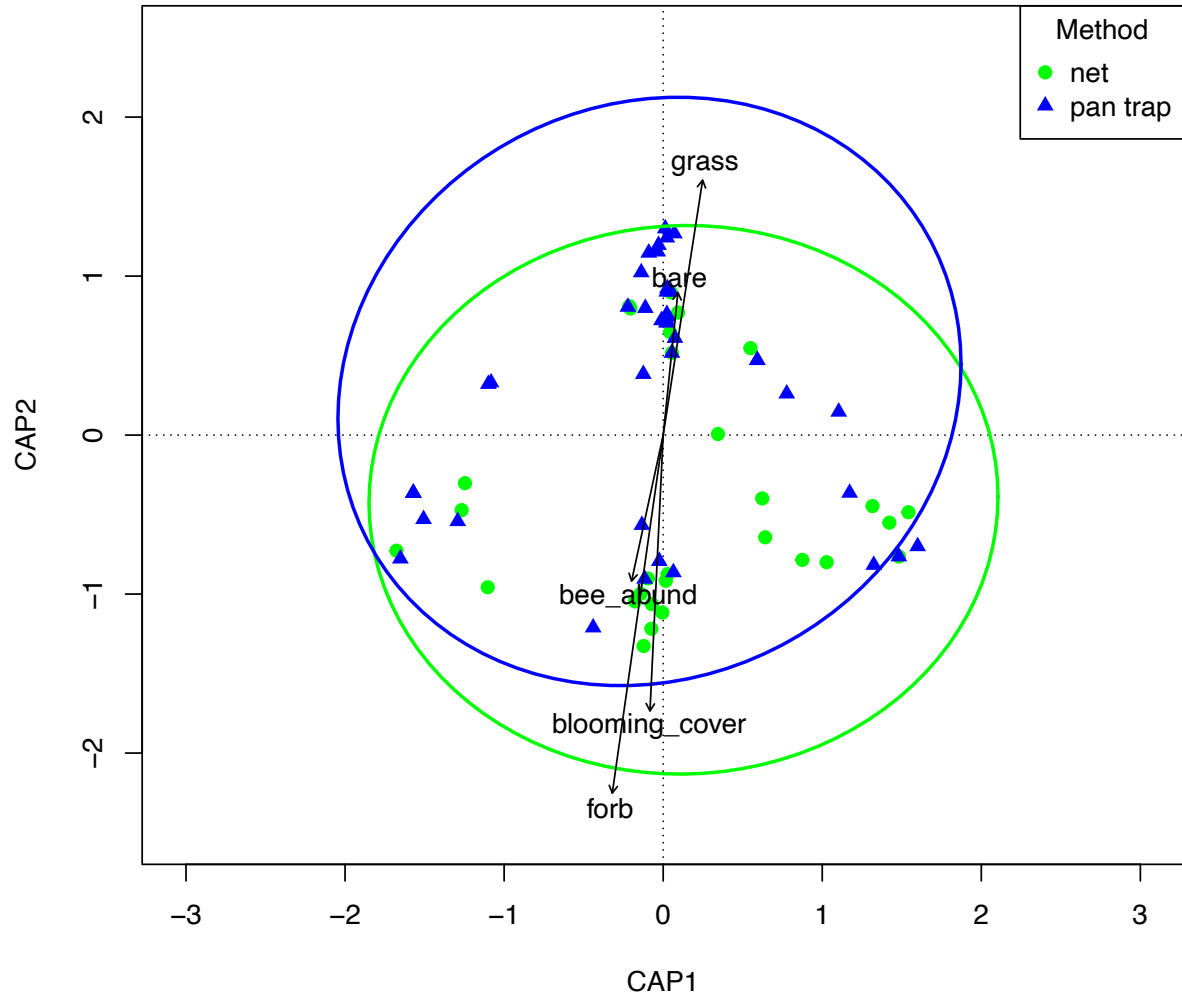


Figure S 7.5 Distance-based redundancy analysis with Bray-Curtis dissimilarity showing the community of bees collected pan traps versus nets. Ellipses represent 95% confidence intervals for each treatment. Vector labels: grass percent ground cover of grasses found in the vegetation transects; bare percent bare ground cover found in the vegetation transects; bee_abund abundance of bees collected in each sample; blooming_cover percent ground cover of blooming forbs found in the vegetation transects; forb percent ground cover of forbs found in the vegetation transects. Sample sizes: net $n=29$; pan trap $n=43$. Figure was created using the ordiplot function from the vegan package in R (Oksanen et al. 2019; R Core Team 2023).

dbRDA of Carabids on Field Edges from Pitfall Traps

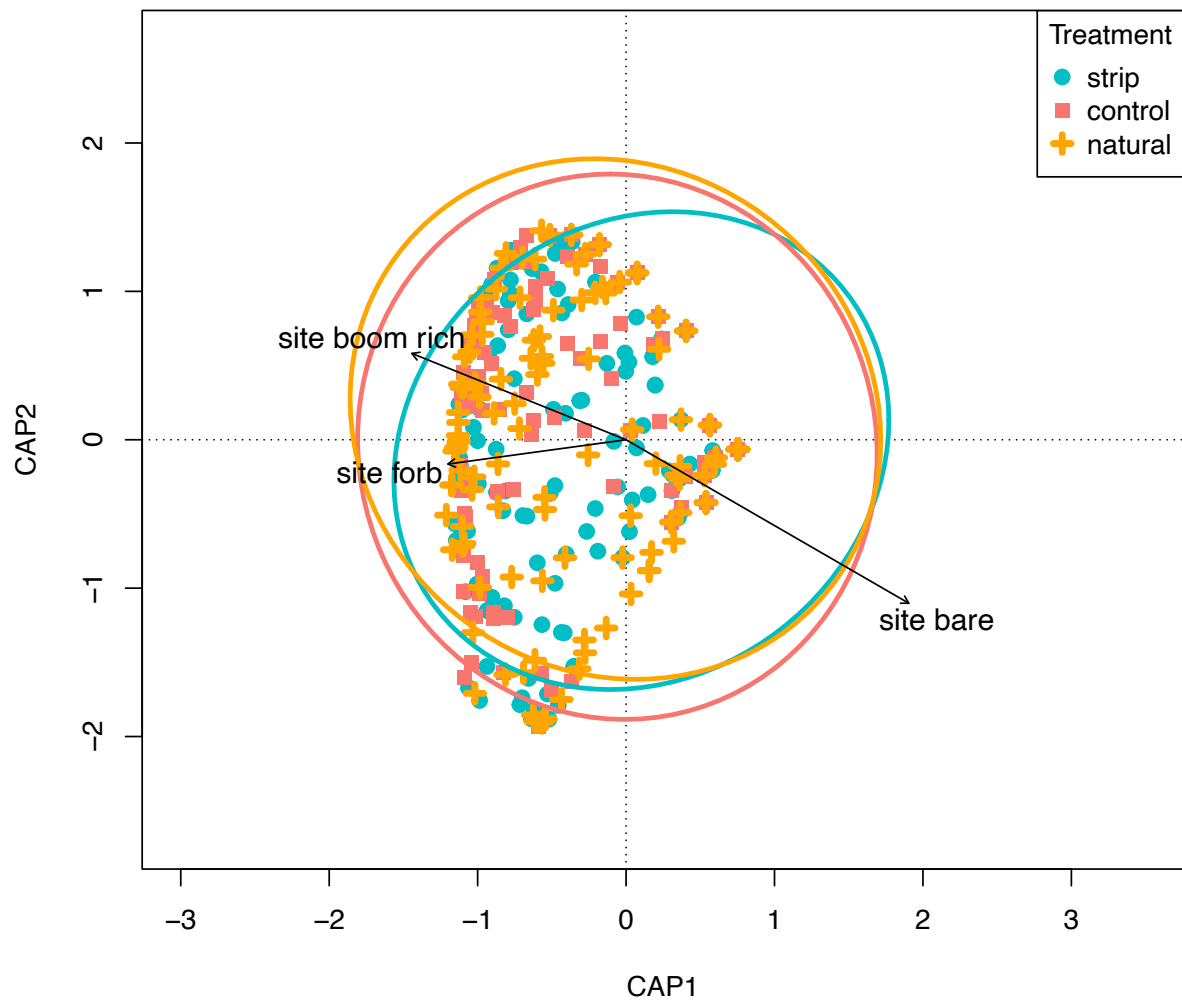


Figure S 7.6 Distance-based redundancy analysis with Bray-Curtis dissimilarity showing the abundance of carabid genera collected with pitfall traps on field edges at sites from three treatments. Treatments included strip locations adjacent to crops with floral strip; control locations adjacent to crops without any enhancements; and natural semi-natural areas without management. Ellipses represent 95% confidence intervals for each treatment. Vectors were included in the plots if they had a significant effect on the ordination ($p < 0.05$) and were calculated using the *envfit* function from the *vegan* package (Oksanen et al. 2019). Vector labels: Site bloom rich average blooming forb richness per quadrat found in the vegetation surveys averaged across the sampling location. Site forb percent of forb cover found in the vegetation surveys averaged across the sampling location. Site bare percent of bare ground cover found in the vegetation surveys averaged across the sampling location. Sample sizes: strip $n=43$; control $n=34$; and natural $n=34$. Figure was created using the *ordiplot* function from the *vegan* package in R (R Core Team 2023; Oksanen et al. 2019).

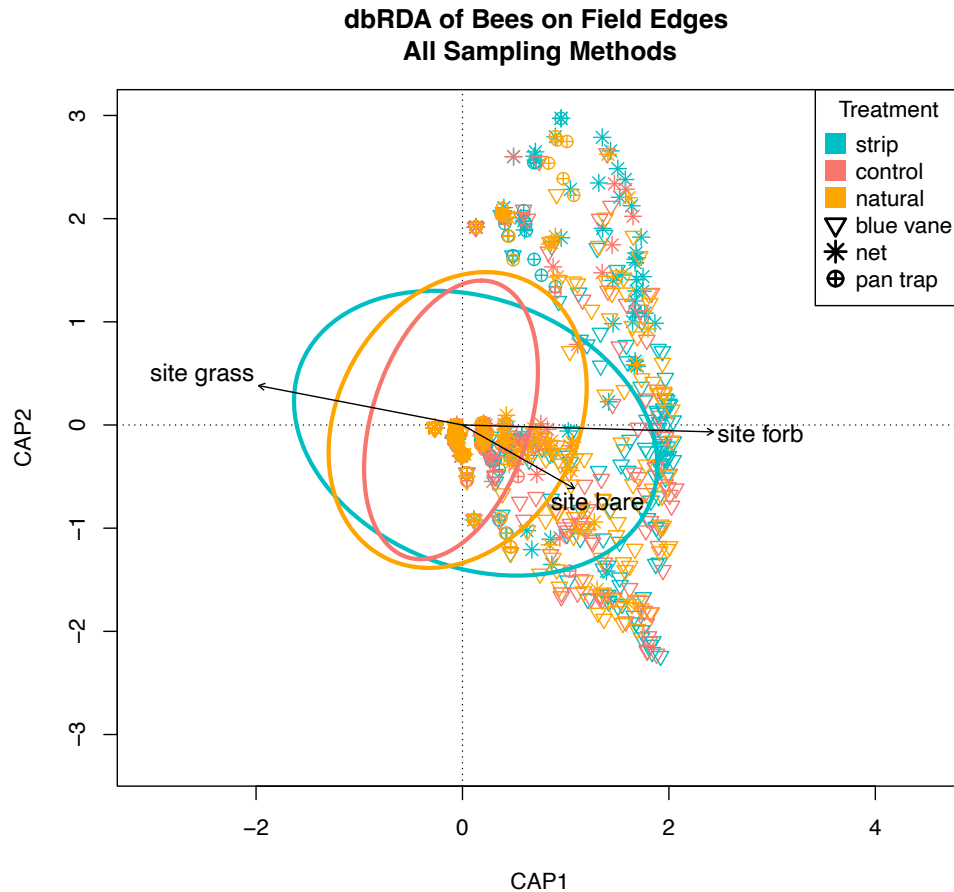


Figure S 7.7 Distance-based redundancy analysis with Bray-Curtis dissimilarity showing the abundance of bees collected with blue vane traps net transects and pan traps on field edges at sites from three treatments. Treatments included strip locations adjacent to crops with floral strip; control locations adjacent to crops without any enhancements; and natural semi-natural areas without management. Ellipses represent 95% confidence intervals for each treatment. Vectors were included in the plots if they had a significant effect on the ordination ($p < 0.05$) and were calculated using the *envfit* function from the vegan package (Oksanen et al. 2019). Vector labels: One vector site bloom rich (0.99956 -0.02953 average blooming forb richness per quadrat found in the vegetation surveys averaged across the sampling location) was removed to increase readability of the plot since it directly overlapped with site forb. Site grass percent of grass cover found in the vegetation surveys averaged across the sampling location. Site bare percent of bare ground cover found in the vegetation surveys averaged across the sampling location. Site forb percent of forb cover found in the vegetation surveys averaged across the sampling location. Sample sizes: control blue vane: $n = 35$; control net: $n = 56$; control pan trap: $n = 47$; natural blue vane: $n = 35$; natural net: $n = 56$; natural pan trap: $n = 48$; strip blue vane: $n = 44$; strip net: $n = 62$; strip pan trap 59. Figure was created using the *ordiplot* function from the vegan package in R (R Core Team 2023; Oksanen et al. 2019).

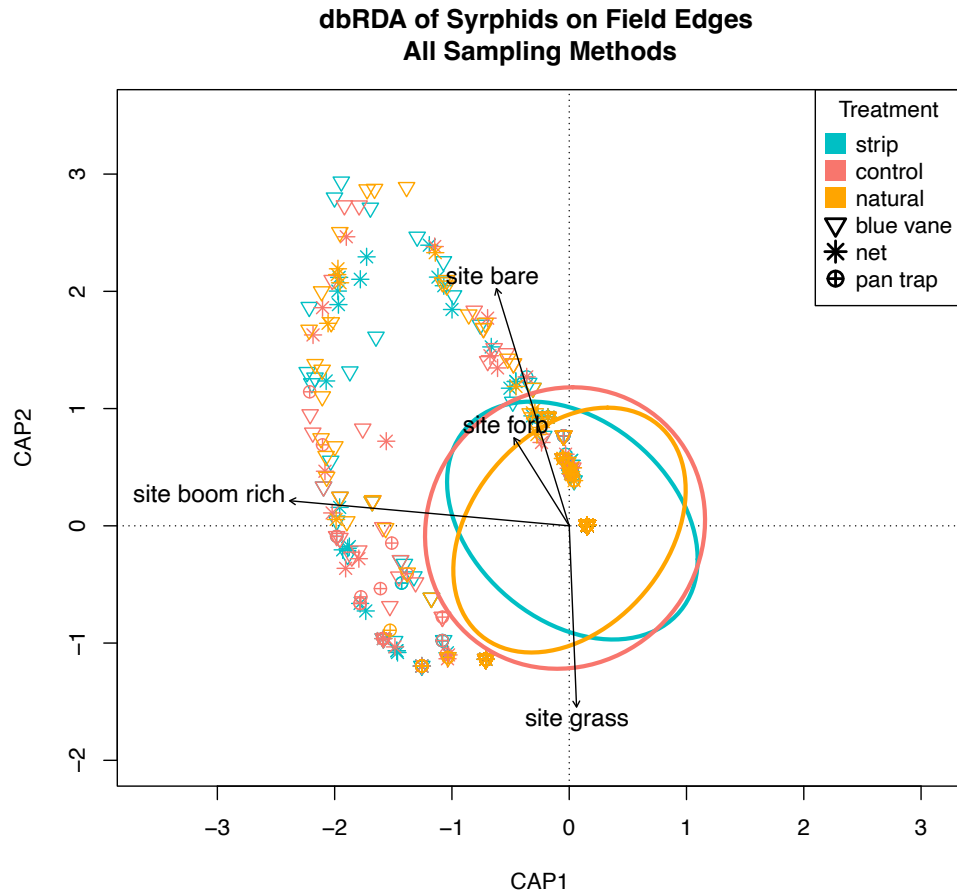


Figure S 7.8 Distance-based redundancy analysis with Bray-Curtis dissimilarity showing the abundance of syrphids collected with blue vane traps net transects and pan traps on field edges at sites from three treatments. Treatments included strip locations adjacent to crops with floral strip; control locations adjacent to crops without any enhancements; and natural semi-natural areas without management. Vectors were included in the plots if they had a significant effect on the ordination ($p < 0.05$) and were calculated using the *envfit* function from the *vegan* package (Oksanen et al. 2019). Vector labels: Site bloom rich average blooming forb richness per quadrat found in the vegetation surveys averaged across the sampling location. Site grass percent of grass cover found in the vegetation surveys averaged across the sampling location. Site bare percent of bare ground cover found in the vegetation surveys averaged across the sampling location. Site forb percent of forb cover found in the vegetation surveys averaged across the sampling location. Sample sizes: control blue vane: $n = 35$; control net: $n = 56$; control pan trap: $n = 47$; natural blue vane: $n = 35$; natural net: $n = 56$; natural pan trap: $n = 48$; strip blue vane: $n = 44$; strip net: $n = 62$; strip pan trap 59. Figure was created using the *ordiplot* function from the *vegan* package in R (R Core Team 2023; Oksanen et al. 2019).

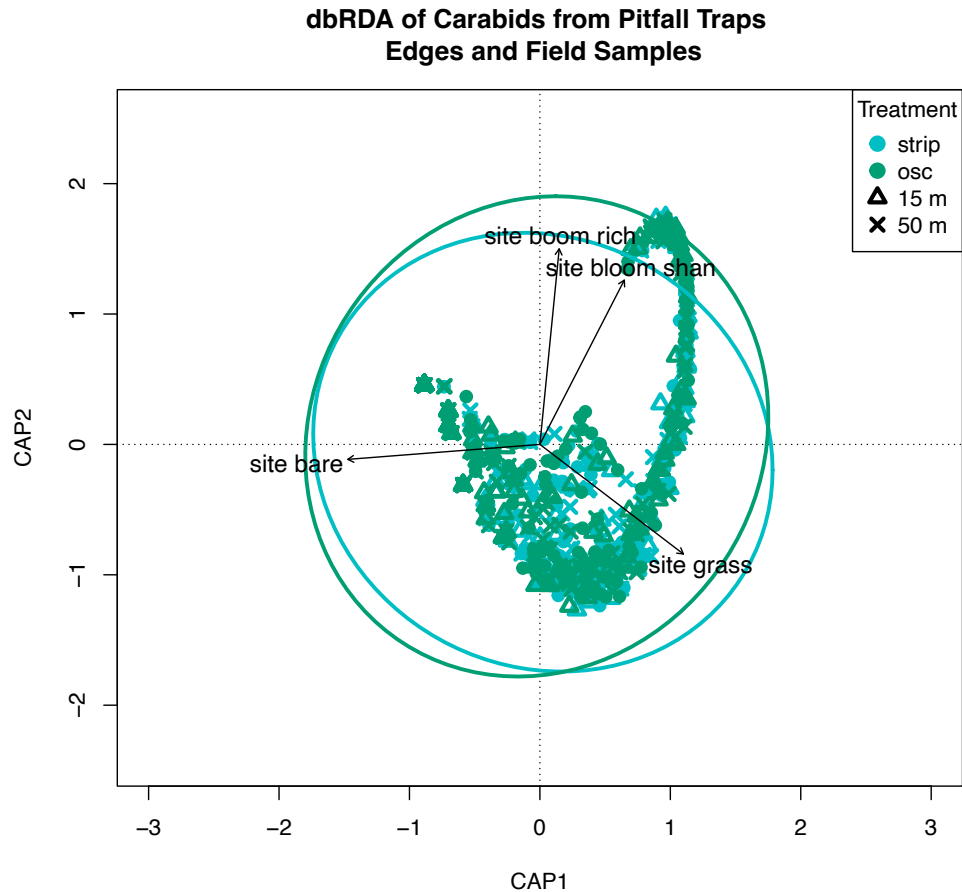


Figure S 7.9 Distance-based redundancy analysis with Bray-Curtis dissimilarity showing the abundance of carabid genera collected with pitfall traps on field edges 15 and 50 m into all sampled crop fields. Treatments included strip sampling locations within strips and directly adjacent to floral strips within an adjacent crop field; OSC sampling locations at the same field as the strips but were located on an unmanaged grassy edge that was at least 150 m away from floral strips. Locations were either within the OSC margin or within crop fields at 15 and 50 m directly in front of the OSC margin. Ellipses represent 95% confidence intervals for each treatment. Vectors were included in the plots if they had a significant effect on the ordination ($p < 0.05$) and were calculated using the *envfit* function from the *vegan* package (Oksanen et al. 2019). Vector labels: Site bare percent of bare ground cover found in the vegetation surveys averaged across the sampling location. Site grass percent of grass cover found in the vegetation surveys averaged across the sampling location. Site bloom rich average species richness of blooming forb species per quadrat found in the vegetation surveys averaged across the sampling location. Site bloom shan average Shannon diversity of blooming forb species per quadrat found in the vegetation surveys averaged across the sampling location. Sample sizes: strip $n=43$; OSC $n=43$. Figure was created using the *ordiplot* function from the *vegan* package in R (R Core Team 2023; Oksanen et al. 2019).

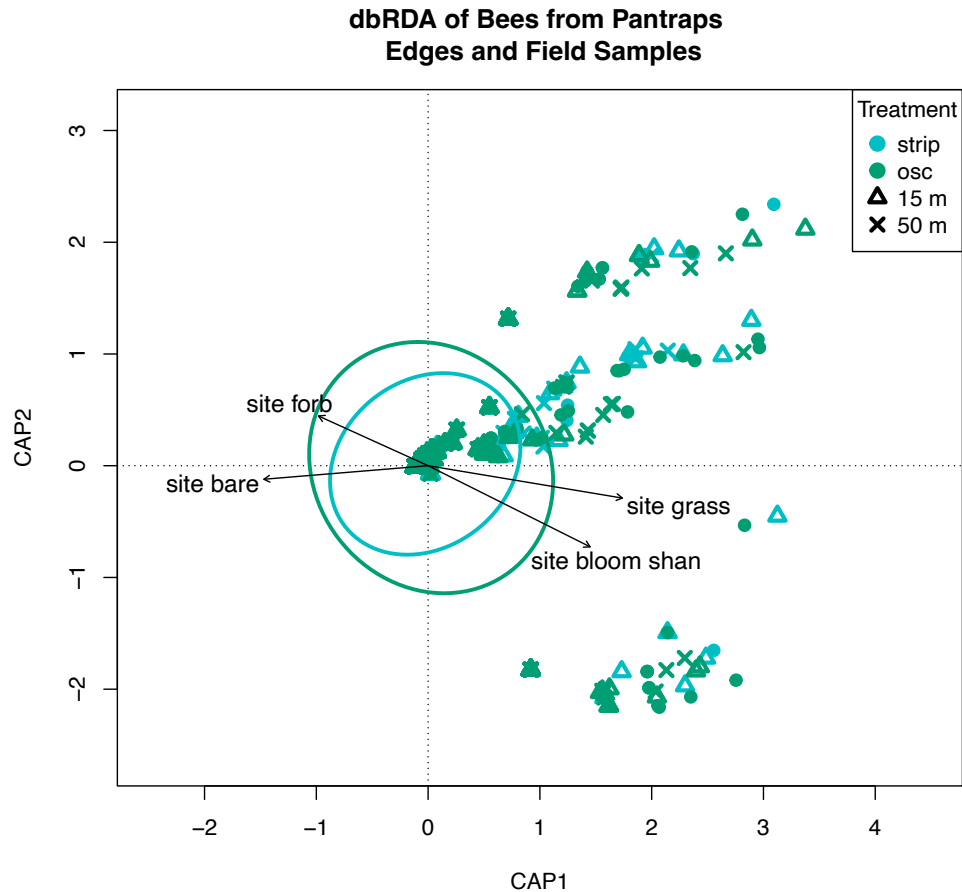


Figure S 7.10 Distance-based redundancy analysis with Bray-Curtis dissimilarity showing the abundance of bee species collected with pan traps on field edges 15 and 50 m into all sampled crop fields. Treatments included strip sampling locations within strips and directly adjacent to floral strips within an adjacent crop field; OSC sampling locations at the same field as the strips but were located on an unmanaged grassy edge that was at least 150 m away from floral strips. Locations were either within the OSC margin or within crop fields at 15 and 50 m directly in front of the OSC margin. Ellipses represent 95% confidence intervals for each treatment. Vectors were included in the plots if they had a significant effect on the ordination ($p < 0.05$) and were calculated using the *envfit* function from the *vegan* package (Oksanen et al. 2019). Vector labels: Site forb percent of forb cover found in the vegetation surveys averaged across the sampling location. Ave wind average wind speed as collected from each site the day of sampling. Max wind average wind speed as collected from each site the day of sampling. Site bare percent of bare ground cover found in the vegetation surveys averaged across the sampling location. Temperature ambient air temperature measured on the day of collection. Site bloom rich average species richness of blooming forb species per quadrat found in the vegetation surveys averaged across the sampling location. Site bloom shan average Shannon diversity of blooming forb species per quadrat found in the vegetation surveys averaged across the sampling location. Site grass percent of grass cover found in the vegetation surveys averaged across the sampling location. Sample sizes: strip $n=59$;

OSC $n=58$. Figure was created using the `ordiplot` function from the `vegan` package in R (R Core Team 2023; Oksanen et al. 2019).

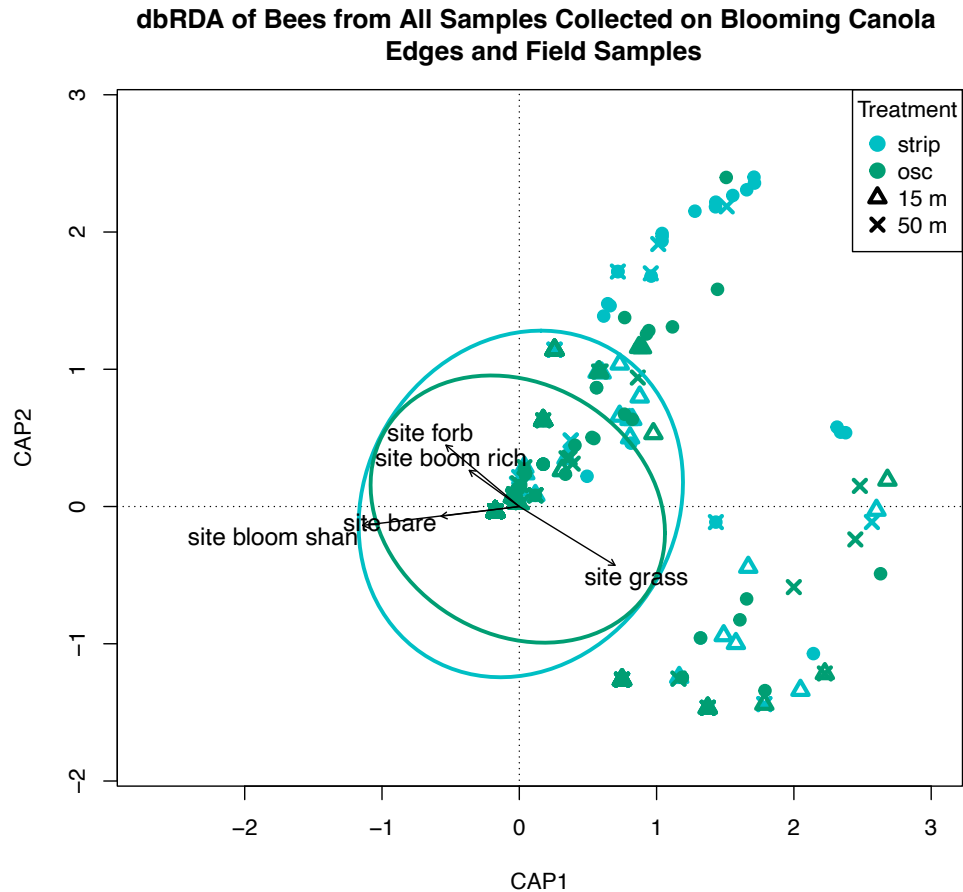


Figure S 7.11 Distance-based redundancy analysis with Bray-Curtis dissimilarity showing the abundance of bee species collected with pan traps and net transects on field edges 15 and 50 m into blooming canola fields. Treatments included strip sampling locations within strips and directly adjacent to floral strips within an adjacent crop field; OSC sampling locations at the same field as the strips but were located on an unmanaged grassy edge that was at least 150 m away from floral strips. Locations were either within the OSC margin or within crop fields at 15 and 50 m directly in front of the OSC margin. Ellipses represent 95% confidence intervals for each treatment. Vectors were included in the plots if they had a significant effect on the ordination ($p < 0.05$) and were calculated using the *envfit* function from the *vegan* package (Oksanen et al. 2019). Vector labels: Site forb percent of forb cover found in the vegetation surveys averaged across the sampling location. Site bare percent of bare ground cover found in the vegetation surveys averaged across the sampling location. Site bloom rich average species richness of blooming forb species per quadrat found in the vegetation surveys averaged across the sampling location. Site bloom shan average Shannon diversity of blooming forb species per quadrat found in the vegetation surveys averaged across the sampling location. Site grass percent of grass cover found in the vegetation surveys averaged across the sampling location. Sample sizes: strip $n=17$; OSC $n=17$. Figure was created using the *ordiplot* function from the *vegan* package in R (R Core Team 2023; Oksanen et al. 2019).

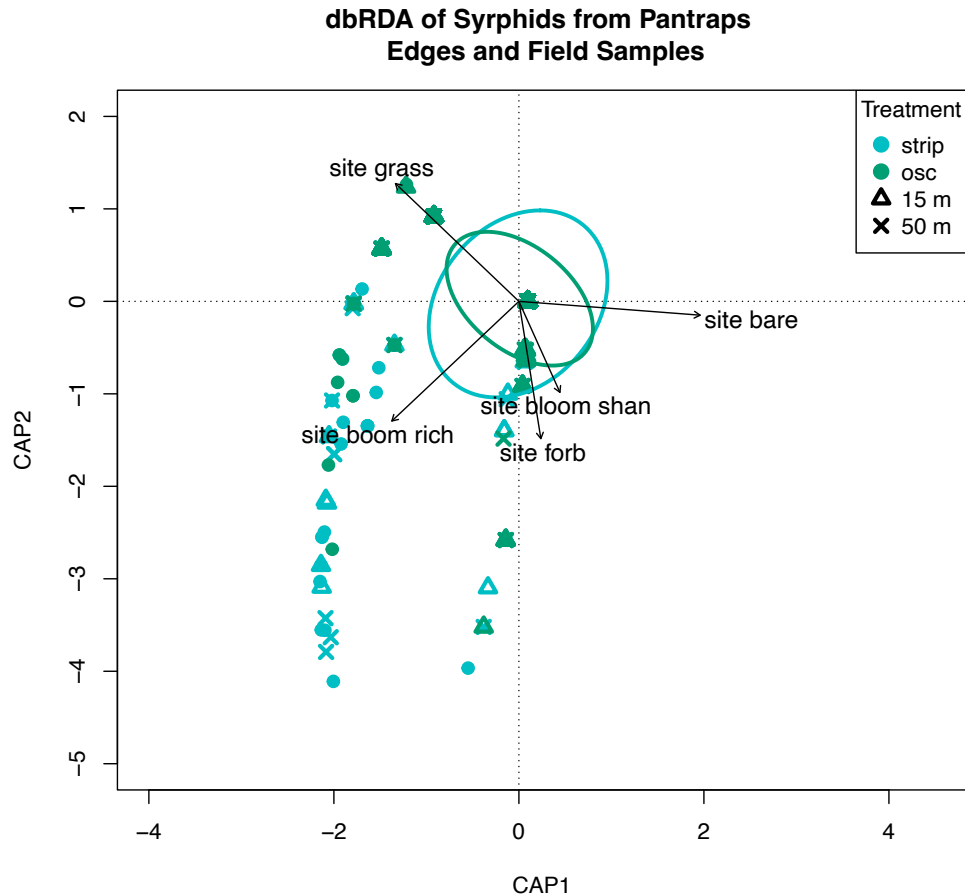


Figure S 7.12 Distance-based redundancy analysis with Bray-Curtis dissimilarity showing the abundance of syrphid species collected with pan traps on field edges 15 and 50 m into all sampled crop fields. Treatments included strip sampling locations within strips and directly adjacent to floral strips within an adjacent crop field; OSC sampling locations at the same field as the strips but were located on an unmanaged grassy edge that was at least 150 m away from floral strips. Locations were either within the OSC margin or within crop fields at 15 and 50 m directly in front of the OSC margin. Ellipses represent 95% confidence intervals for each treatment. Vectors were included in the plots if they had a significant effect on the ordination ($p < 0.05$) and were calculated using the *envfit* function from the *vegan* package (Oksanen et al. 2019). Vector labels: Site grass percent of grass cover found in the vegetation surveys averaged across the sampling location. Site bloom rich average species richness of blooming forb species per quadrat found in the vegetation surveys averaged across the sampling location. Site bloom shan average Shannon diversity of blooming forb species per quadrat found in the vegetation surveys averaged across the sampling location. Site forb percent of forb cover found in the vegetation surveys averaged across the sampling location. Ave wind average wind speed as collected from each site the day of sampling. Max wind average wind speed as collected from each site the day of sampling. Site bare percent of bare ground cover found in the vegetation surveys averaged across the sampling location. Sample sizes: strip $n=59$; OSC $n=58$. Figure was created using the *ordiplot* function from the *vegan* package in R (R Core Team 2023; Oksanen et al. 2019).

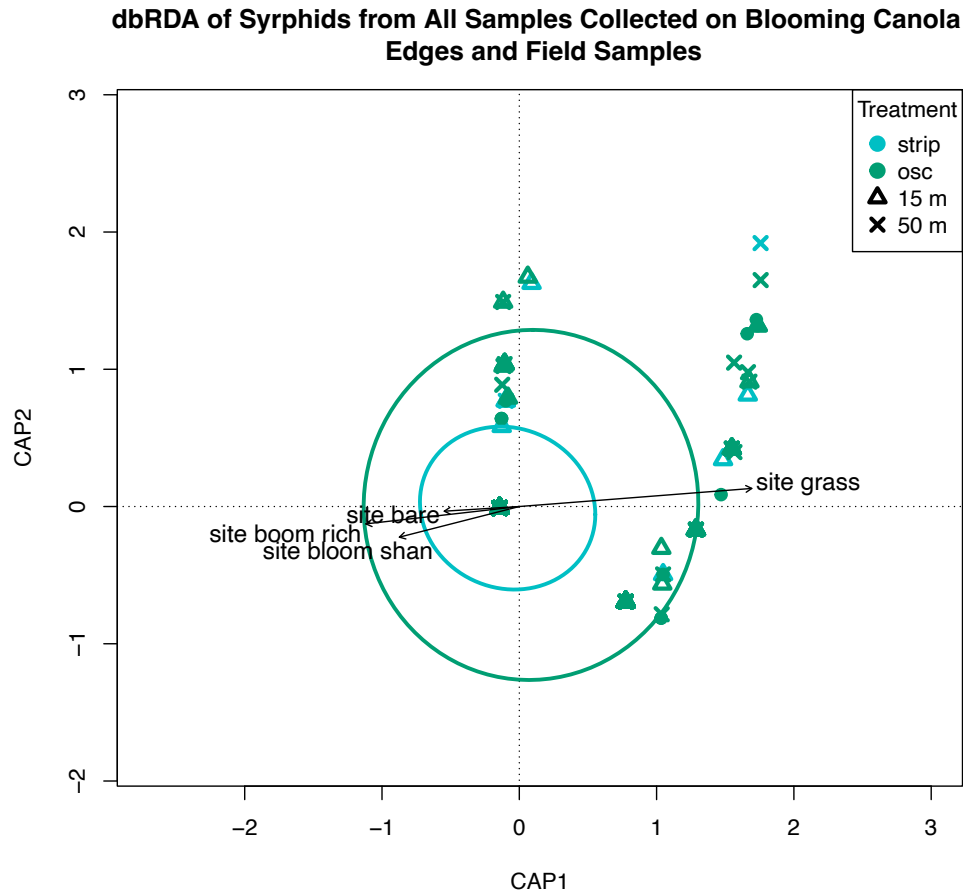


Figure S 7.13 Distance-based redundancy analysis with Bray-Curtis dissimilarity showing the abundance of syrphid species collected with pan traps and net transects on field edges 15 and 50 m into blooming canola fields. Treatments included strip sampling locations within strips and directly adjacent to floral strips within an adjacent crop field; OSC sampling locations at the same field as the strips but were located on an unmanaged grassy edge that was at least 150 m away from floral strips. Locations were either within the OSC margin or within crop fields at 15 and 50 m directly in front of the OSC margin. Ellipses represent 95% confidence intervals for each treatment. Vectors were included in the plots if they had a significant effect on the ordination ($p < 0.05$) and were calculated using the *envfit* function from the *vegan* package (Oksanen et al. 2019). One vector was removed from the final plot to increase readability-Site forb. Site forb had a similar vector direction and length (Cap1: -0.76696 Cap2: 0.64169) to site bloom rich. Vector labels: Site forb percent of forb cover found in the vegetation surveys averaged across the sampling location. Site bare percent of bare ground cover found in the vegetation surveys averaged across the sampling location. Site bloom rich average species richness of blooming forb species per quadrat found in the vegetation surveys averaged across the sampling location. Site bloom shan average Shannon diversity of blooming forb species per quadrat found in the vegetation surveys averaged across the sampling location. Site grass percent of grass cover found in the vegetation surveys averaged across the sampling location. Sample sizes: strip $n=17$; OSC $n=17$. Figure was created using the *ordiplot* function from the *vegan* package in R (R Core Team 2023; Oksanen et al. 2019).

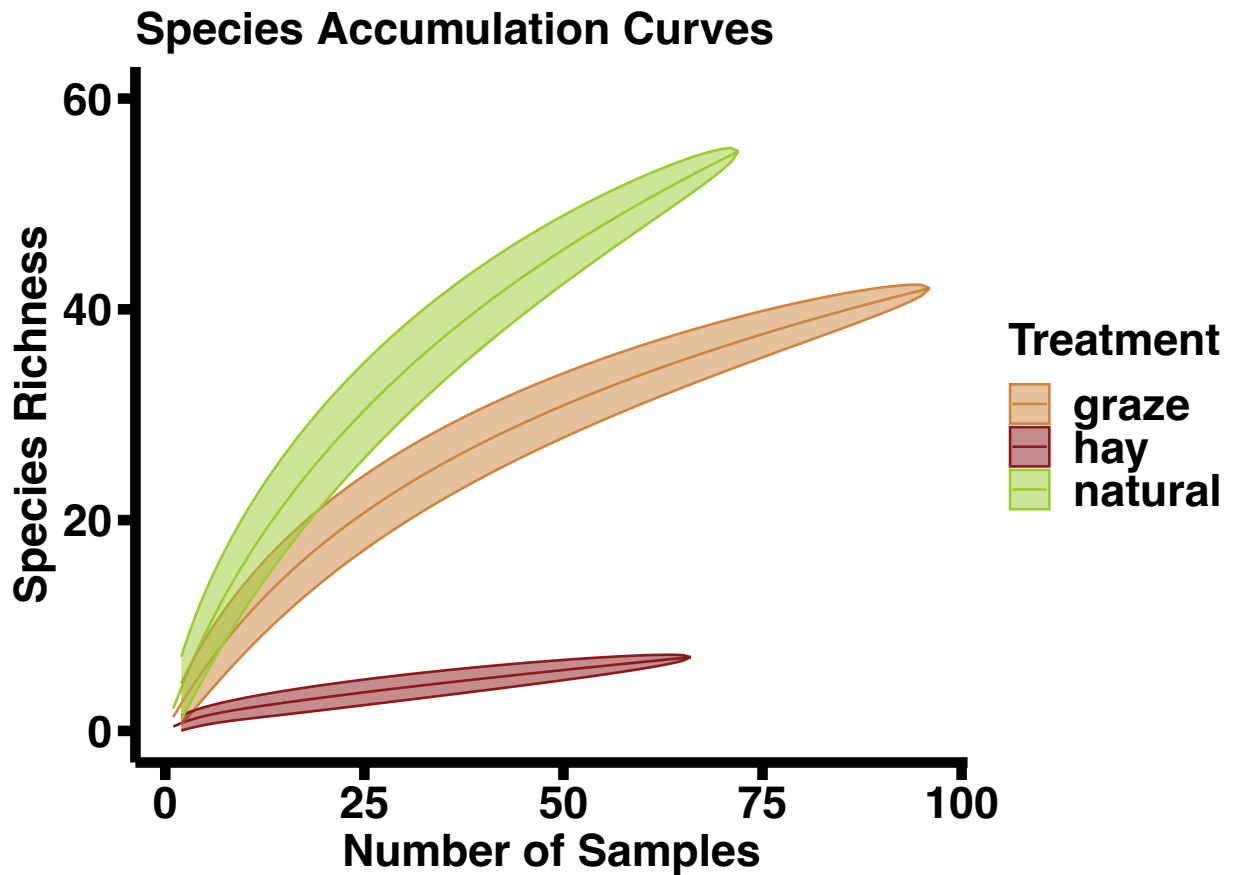


Figure S 7.14 Species accumulation curve of three treatments of sites sampled in Manitoba Canada during the summers of 2020 – 2022.0 Treatments included graze; locations that are rotationally grazed by cattle natural; semi-natural sites that have not been managed for at least 5 years and hay; fields that have been planted with an enhanced hay mixture.