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## Executive Summary

Beginning in 2012, Pollinator Partnership initiated monitoring and assessment of newly installed Pollinator Habitat Practice (PHP) seed mixes. The overall goal of this study was to evaluate and establish the supportive role that PHP provided to honey bees and native bees. More specific goals of this research project included: increasing the understanding of the relationship between honey bees, native bees, and selected plants established under different Conservation Reserve Program (CRP) covers; quantifying the costs and benefits of the seed mixes used in CRP covers to honey bee productivity and native bee occurrence; and developing a means by which FSA can readily assess CRP effects on pollinator populations and services as needed. Research and monitoring occurred in two commodity crop systems, wheat in Montana and Washington, and corn in Iowa and Nebraska. Responses in the pollinator community measured during this five-year study period included: honey bee productivity associated with each landscape type (measured through hive weight gain); native bee productivity associated with each landscape (measured through tube nest occupancy); and honey bee and native bee support provided by each landscape type (measured through floral visitation assessments and plant-insect interaction networks). Cost-benefit models for PHP and CRP landscapes were developed using enrollment costs and resulting economic and ecological benefits.

PHP provided increased value to honey bees foraging on these landscapes measured in increased hive weight gain, and provided native bees with increased community support, as indicated by recorded patterns of richness and abundance. Results indicated a significant categorical difference between the pollinator support value (seen through greater hive weight gain) in PHP compared to standard CRP ( $p < 0.001$ ). Honey bee productivity, in terms of honey produced, did not change with increasing enrollment size of CRP or PHP. However, native bee abundance and richness was significantly correlated with increasing PHP enrollment size. These results, and other emerging work in seed mix optimization for pollinator community support, suggest that specific bee-plant interactions are driving the trends observed. Some plant species provided a greater value to honey bees and the native bee community and their inclusion in PHP seed mixes is primarily responsible for the pollinator benefits seen.

PHP is more costly than CRP. Mean PHP per acre enrollment costs were \$264.10 more costly per acre than CRP (30.95 vs. 295.05). Fewer acres are enrolled in PHP; however, these areas, on average, cost 9.5 times more than CRP acres, and are embedded within CRP land. PHP costs are higher due to high seed mix costs and Sign-up Incentive Payments (SIP) associated with the practice. This increased cost corresponded to significantly higher honey production and native bee support benefits. PHP land provided 33.17 more pounds of honey per colony, 37% more bees, and a 70% increase in bee taxa compared to samples on CRP land. The cost of honey production can be further assessed based on each dollar spent on CRP or PHP enrollment; one CRP enrollment dollar accounts for 0.03 pounds of honey and one PHP enrollment dollar accounts for 0.07 pounds of honey, 2.3 times the amount of honey. It should be noted that PHP land is embedded within CRP land and it is not possible to calculate the benefit per acre or how bees were using the landscape. It is clear however that PHP enhancements substantially increased hive honey production.

Significant correlations were observed between increasing acreage of PHP seedings, and increasing native bee abundance and richness. An area-based benefit is also evident, with each increasing acre of PHP seeding corresponding to 14.88 more individual native bee visits to flowers observed and 1.02 more taxa. Although this is an area-based response, it is likely driven by the increased abundance of preferred native bee forage plants present within the landscape. Plant-insect interaction networks for each CRP and PHP landscape assessed in Iowa and Nebraska from data collected through the 2015-2016 season characterize these relationships. Native plants included in PHP mixes with the highest plant value index, meaning those that supported the highest number of bees and bee taxa, were Canada Goldenrod (*Solidago canadensis*), Oxeye False Sunflower (*Heliopsis helianthoides*), Wild Bergamot (*Monarda fistulosa*), Purple Aster (*Symphotrichum* sp.), Purple Prairie Clover (*Dalea purpurea*), Yellow Coneflower (*Ratibida pinnata*), Black-eyed Susan (*Rudbeckia hirta*), and Common Sunflower (*Helianthus annuus*). Non-native plants including White Sweet Clover (*Melilotus albus*) and Yellow Sweet Clover (*Melilotus officinalis*) were also noted to provide significant support to pollinators, but these were not actively seeded in PHP.

The enhanced value of PHP mixes to honey bees and native bees is supported by our research. PHP mixes should continue to be promoted as a highly effective strategy for providing resources for honey bees and enhancing native bee communities. Our results indicate an overall benefit provided to both honey bees and native bees that is driven by the plant community characteristics of PHP. Optimizing seed mixes and encouraging an increase in the acreage enrolled in PHP are valuable strategies for improving honey bee productivity and biodiversity given the benefits measured in this study.

A key recommendation from this research is ensuring the inclusion of specific high-value pollinator plant species in PHP mixes which are likely responsible for driving the managed and native bee benefits. In areas with PHP enrollment options where interaction networks and assessments of pollinator support have not been conducted, likely the majority of regions, we recommend continuing with high diversity plant mixes, increasing the odds of including 'magnet', high-value plant species, while at the same time conducting plant-bee interaction assessments to validate plant species function. We recommend additional research into the development of plant-pollinator interaction networks through field assessment for all PHP mixes to complement this work and work that has begun to outline patterns of plant use in the northern Great Plains.

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## Project Background

Conservation Reserve Program (CRP) lands, although originally designated to address issues of soil erosion, have evolved to include significant wildlife conservation benefits. Species benefitting from CRP lands include mammals such as bison (Seefeldt et al. 2010), pronghorn (Griffin 1991), mule deer (Kamler et al. 2001), and swift foxes (Kamler et al. 2003); birds such as sage grouse (Schroeder and Vander Haegen 2006), bobolinks (Bollinger and Gavin 1889), and other song birds (Bertherlsen and Smith 1995; Best et al. 1997; Hull et al. 1996); reptiles and amphibians (McIntyre 2003); and invertebrates (McIntyre and Thompson 2003, Burger et al. 1993), including pollinators (Davros et al. 2006, Reeder et al. 2005; Reeder et al. 2005; Ries and Debinski 2001).

The use of CRP lands for managed pollinator support (honey bee forage) has been particularly established in the northern plains, with beekeepers pasturing colonies on CRP lands for summer rest and honey production. More recently, the Honey Bee Habitat Initiative was established throughout five Midwestern states to enhance the value of USDA conservation lands to honey bee support (FSA 2015). The Honey Bee Habitat Initiative provides cost-share incentives for mid-contract management to include the seeding of nectar plants preferred by honey bees and that are known to benefit honey production.

Although support of pollinators on CRP lands has been documented prior to the initiation of the pollinator habitat conservation practice (see Davros et al. 2006, Reeder et al. 2005; Reeder et al. 2005; Ries and Debinski 2001); at the time of the development of this report, published assessments of CRP benefit to pollinators were not available. This assessment and report represents the first review of the new practice installed during sign-up 39.

### **Conservation Practice 42 – Pollinator Habitat Establishment Practices**

Conservation Practice 42 (CP-42), also known as the Pollinator Habitat Establishment Practice (PHP), provides a specific set of guidelines offering additional qualification criteria to CRP enrollment. Land enrolled in PHP usually is embedded within larger CRP lands. Therefore, PHP lands provide enhancement for pollinators (and potentially other wildlife) on CRP land. PHP criteria have been created to increase the value of CRP lands to honey bees and native pollinators, based on our current understanding of the needs of bees within agroecosystems. For an enrollment to be considered eligible for PHP, guidelines specify the following:

[https://www.fsa.usda.gov/Assets/USDA-FSA-Public/usdafiles/FactSheets/2015/CRPPProgramsandInitiatives/Pratice\\_CP42\\_Pollinator\\_Habitat.pdf](https://www.fsa.usda.gov/Assets/USDA-FSA-Public/usdafiles/FactSheets/2015/CRPPProgramsandInitiatives/Pratice_CP42_Pollinator_Habitat.pdf)  
[https://www.fsa.usda.gov/Internet/FSA\\_File/cp42\\_habitat.pdf](https://www.fsa.usda.gov/Internet/FSA_File/cp42_habitat.pdf)  
[https://www.fsa.usda.gov/Internet/FSA\\_File/pollinator\\_fact\\_sht.pdf](https://www.fsa.usda.gov/Internet/FSA_File/pollinator_fact_sht.pdf)):

Participants must:

- Have owned or operated the land for more than 12 months prior to program sign-up
- Be in control of the land for the length of the contract
- Meet USDA payment eligibility provisions

Practice Requirements:

- Habitat areas must be at least 0.5 acres each
- Noxious weeds and other undesirable plants, insects, and pests should be controlled



- Must meet additional technical requirements (If not planted in whole fields, block plantings are preferred over strips)
  - The land must be cropland and meet cropping history requirements as specified in the Code of Federal Regulations.
  - Land currently enrolled in CRP may be re-offered for enrollment into CP42 if the land enrolled in CRP is in the last year of the CRP-1 contract.
  - Whole fields may be enrolled. If not planted in whole fields, block plantings are preferred over strips. If planted in strips, each strip must be a minimum of 20 feet wide.
  - Contracts for CP42 practices must have a duration of 10 years.
  - Grasses seeded in this practice must be native. Although native species are encouraged, beneficial introduced flowering plants (e.g., alfalfa and clover) may be part of the seeding mix if approved by USDA conservation planners
  - Seeding mixes generally require a minimum of nine species of pollinator-friendly wild-flowers, legumes, and/or shrubs, but states with arid areas may submit alternative standards
  - The pollinator practice must be maintained for the duration of the contract.

Financial incentives for participating in PHP include:

- 10 years of annual rental payments
- Payment covering 50% of the eligible costs of establishing the pollinator practice
- Sign-up Incentive Payment (SIP) up to \$150/acre if enrolled in continuous sign-up
- 50% Cost-Share Payment for mid-contract management

***Note that at the time this report was finalized (March 2018) CRP enrollment was frozen and policies were under review.***

Initial registration during the first year of PHP availability (sign-up 39) resulted in 41,231 acres enrolled with this practice across the United States, representing 0.1% of the 31,290,459 total national acres enrolled in CRP (FSA 2011). States leading PHP enrollment included Colorado (19,856 acres), Idaho (3,360 acres), and Nebraska (3,031 acres). States selected for study in this review represented high PHP acreages: Nebraska (3,031 acres), Iowa (1,731 acres), Montana (1,720 acres), and Washington (1,145 acres) were selected for study in 2011.

PHP is a relatively new practice that can incur an elevated cost due to the pricing of native wildflower seeds required for the mix. Seed mixes are designed by NRCS and other conservation partners (including The Xerces Society and Pheasants Forever), based on natural history confirming pollinator use and the incorporation of plant species deemed supportive by pollinator syndromes or other available ecological data. Local availability of native wildflower seed also plays a role in seed mix development. The diversity and regional patterns of the native plants and the native pollinator community are so great that there are broad areas of the United States in which vetted, peer-reviewed and published data validating plant suitability are unavailable. This study of PHP affords Pollinator Partnership the opportunity to acquire field data to validate the currently used and proposed plant species. With this study we evaluate the overall functioning of PHP compared to other established CRP lands (primarily CP1 and CP2) for increased pollinator support.

We further investigate the characteristics of PHP seed mixes, aiming to evaluate the pollinator relationships with each plant component.

The seeding of high diversity pollinator mixes (mixes containing over 25 native plant species) is widely practiced. PHP enrollments in Nebraska and Iowa in particular include high diversity mixes promoted by local conservation groups (i.e., Pheasants Forever). In other states, enrollments are characterized as being at or near minimum requirements (i.e., Montana and Washington), due to a number of factors including limited supply of commercially available native seed, both in volume and variety, and dry environments that limit the number of candidate species for the ecological region in question.

Striving for increased diversity in ecosystems is considered the default approach, as high diversity systems are most commonly characterized by higher resilience and productivity (Potts et al. 2010; Thebault and Fontaine 2010; Carvalheiro 2011; Burkle et al. 2013); both characteristics that are desired in pollinator communities. However, research into plant-pollinator interactions (outlined in more detail in subsequent sections of this report) does not indicate a universal trend of consistently increasing ecosystem function with increasing plant and pollinator diversity (Flemming et al. 2001; Memmott et al. 2004; Ebeling et al. 2008). Rather, saturation and redundancy are seen at intermediate values of diversity (see trends emerging in Otto et al. 2016 as well as personal communications N. Williams, UCD). These values vary from system to system. A minimum of 9 species in the seed mix may very well be sufficient to achieve the full pollinator support goal of PHP. The question remains as to which nine species are to be selected that are going to provide the best support to the widest range of pollinators.

At the landscape level there also is the question of leveraging conservation dollars to increase overall enrollment acres. Would we expect a greater benefit to pollinators for similar cost outlays if more conservative seed mixes using minimum criteria were applied, at a greater number of locations? Or, would there be more benefit from using higher diversity mixes, resulting in fewer sites that support more species? This question has parallels to the SLOSS (single large, or several small) debate in conservation area management. Are species better off with single large areas, or with small parcels covering a larger area scattered throughout the landscape?

An additional factor being considered in this review is the potential to optimize FSA conservation dollars for pollinator benefits. PHP seed mixes currently are significantly more costly than many other conservation practices. This higher cost is attributed to the costs of certain native plant species. Since the initial rollout of the program, seed mix costs have decreased as demand and production align more closely, however, certain native plant seed will remain costly due to production limitations. Results indicating that fewer plant species support sufficient bee communities and productivity could liberate FSA funds for additional enrollments. A basic understanding of the value that each component of a PHP mix provides to the native pollinator and managed pollinator community also provides relevant information by which to assess and prescribe conservation mixes that optimize conservation funds. Seed mix costs potentially could be reduced by incorporating non-native (non-invasive) plants that provide value to pollinators. However, non-native have not evolved with the local pollinator community. There is indication that non-native plants can support an abundance of pollinators, but may not support as high a species or

functional diversity as native plant assemblages (e.g., Morandin and Kremen 2013a, b; Grass et al. 2014; Williams et al. 2015).

### **Characteristics of bees and impacts for their conservation and management**

Bees are broadly considered the most ecologically valuable pollinators, and this characteristic is attributed to their inherent need to collect pollen for reproduction, likening them to high-frequency and high-fidelity plant visits that result in pollination. The link that bees have with their critical feeding habitat, and their ability to fly, thereby making them less subject to some forms of minor landscape fragmentation that impact terrestrial organisms, can aid in the design of conservation habitat. The scale at which fragmentation impacts bee foraging varies between functional groups and is related to retaining reproductive output within their optimal foraging ranges (Gathmann and Tscharrntke 2002; Steffan-Dewenter et al. 2002; Heinrich 1979; Greenleaf et al. 2007). Honey bees can forage efficiently within a 2km radius; bumble bees and other large bees forage efficiently within 1 km of their nest (Steffan-Dewenter et al. 2002). Medium sized bees, such as leafcutter and mason bees, are thought to forage effectively up to 300 meters, and smaller bees, such as sweat bees and other small bees, forage efficiently up to 200 meters (Gathmann and Tscharrntke 2002; Greenleaf et al. 2007). Dispersal and colonization distances likely are much larger, allowing bees to colonize newly created habitat patches. Land use intensification and fragmentation within natural landscapes are a leading cause of pollinator decline (Winfree et al. 2007; McIntyre and Hostetler 2001). Within agroecosystems, land use intensification and fragmentation are drivers of not only pollinator decline, but declines in crop pollination services (Kremen et al. 2006; Kremen et al. 2004). Theoretically, implementing conservation actions that provide habitat patches connected within maximum foraging ranges would serve to reconnect, at least partially, landscapes needed by pollinators.

We know from habitat construction efforts, as well as monitoring of highly modified agroecosystems, that native bees are finding refuge within these landscapes (Hall et al. 2017; Cane et al. 2006; Winfree et al. 2009). Newly created or enhanced habitats are quickly used or colonized by native bees, indicating source populations within the agricultural matrix or nearby natural habitat features (Jha et al. 2009; Tommasi et al. 2004; Frankie et al. 2009; Matteson et al. 2008). PHP landscapes should therefore provide the existing native bee community with increased feeding and nesting opportunities, and populations should be expected to increase. In this study, we assess and quantify benefits of PHP seeding to pollinators. The data collected in this study do not allow for questions on population growth be addressed. Long-term ecological studies of pollinator trends in agroecosystems are needed for this.

Bees are able to forage on resources that are distant from their nest site. In natural, undisturbed landscapes, bees search for preferred or ideal forage within a spatiotemporal matrix that contains many gaps in food availability. Natural gaps in food availability include the spatial distribution of plants as well as seasonal patterns in bloom. And therefore, bees are adapted to seek food in an unstable environment. The ability of bees to find and locate new sources of forage has helped them persist in seasonally changing and disturbed landscapes. Undoubtedly anthropogenic factors have increased the fragmentation experienced by bees; however, their biology suggests that habitat support efforts should be successful.

*If you build it they will come* -- this has been a common statement of practice for bee conservation and management. If bees are present in a landscape, the presence of a used, preferred, or acceptable resource should draw usage and colonization. The basic behavior of scanning landscapes and searching for food, increases the likelihood that bees present in a landscape will encounter and use habitat that has been provided to support them. With this in mind, strategies whereby habitat has been constructed, augmented, or enhanced are more often than not successful in attracting bees.

### **Benefits seen in agro-ecosystems**

It generally has been established that the presence of diverse native plantings near to agricultural areas corresponds with an increase in pollinator community support and pollination services (Lewis 1969, Williams et al. 2015). Crops grown on farms near to natural areas that provide sources of pollinators have greater yield attributed to increased visitation from a diverse pollinator community (Kremen 2007). Correspondingly, hedgerows or other semi-natural habitats planted in agricultural areas attract and export a greater abundance and diversity of pollinators to adjacent agricultural lands (Morandin and Kremen 2013b). The overall benefit of these floral enhancements has been established in productive agricultural systems, with increased yield seen in canola (Morandin and Winston 2006), sunflower (Greenleaf and Kremen 2006), blueberry (Blaauw and Isaacs 2014; Gibbs et al. 2016) and other crops.

Assessments of pollinator conservation seed mixes in natural areas, conservation easements, and other non-productive landscapes have been more limited, primarily due to a lack of direct economic connections and limited project budgets. Pollinator habitats established primarily for conservation purposes have largely not been monitored for their function. Conservation dollars have been focused on the establishment of habitat, not for monitoring and assessment. In instances where habitat enhancements have been made to landscapes, there is a clear indication that seeding with locally native species enhances pollinator communities, resulting in an increase in abundance and species richness. Monitored restorations of landfills (Tarrant et al. 2021; Kutby 2013) and degraded landscapes (Galea et al. 2016) provide evidence that restoration seed mixes attract a diverse pollinator community, often greater than is seen at nearby natural landscapes that have not had any management interventions. This trend is due to two points, 1. Light disturbance on managed lands that can lead to increased opportunity seed banks to respond, and 2. The purposeful installation of species that are known to attract a large amount of pollinators, which are not always the most abundant in natural lands. Efforts to naturalize managed roadsides and utility corridors also provide supportive data indicating seeding with native flowers increases abundance and richness of pollinator species (Ries et al. 2001, Larsen et al. 2010, Hopwood 2008, Hopwood et al. 2010). Further supporting this concept are data indicating greater pollinator use and presence at habitat gardens established in rural, suburban, and urban landscapes (Frankie et al. 2009). While data on landscape-level and larger-scale seeding is highly limited, trends seen in other habitat restoration efforts, and in on-farm enhancements strongly suggest clear conservation benefits from seeding native plants for pollinators. For example, native bees were shown to prefer native to introduced plant species in agricultural hedgerows, with both greater abundance and diversity of bees on the native plants, controlling for floral cover (Morandin and Kremen 2013a). Further, 40% of native bee species (20 of 50 species) were only found at restored hedgerow sites with native plants, not at nearby weedy edge sites with non-native plants, and greater abundance of uncommon bee species

on native flowers than on non-native flowers showed that the native flowers were important for supporting rare species (Morandin and Kremen 2013b). Further, lack of diverse, and abundant floral resources is thought to be a major contributor to poor honey bee colony health (Vaudo et al. 2015). Enhancement of CRP lands with more floral resources will provide a benefit to managed honey bees colonies.

Optimization of conservation practices is of particular interest due to the costs associated with native seed mixes. Although seed mixes have come down in price, and will likely continue to with increasing demand, there is interest in improving the species mix. Studies of plant-pollinator communities in multiple systems indicate an imbalance in species usage. This trend is being formalized with recent computational approaches to plant-pollinator network evaluation, but also is evident in natural history assessments. Keystone, highly attractive, or magnet species are common in natural and modified systems; these plant species are visited by a greater proportion of the pollinator community and at a higher frequency. The relationship of native flowering plants to their pollinator visitors is seldom one-to-one. Recognizing that pollinator communities are complex and we currently do not have an account of all systems, and that many ecologically valuable plant-pollinator relationships may be monolectic, the use of exclusively magnet species can potentially exclude the conservation of rare species. However, a focus on promoting the use of magnet species that are more affordable, more accessible, or more abundant in conservation mixes is a sounds strategy for optimizing conservation dollars.

### **Benefits of CRP to pollinators**

There is very little peer-reviewed research on measurable impacts that CRP landscapes have on bees. The majority of existing work focuses on resource availability to honey bees, and takes the form of large-scale spatial analysis outlining habitat loss (Wright and Wimberly 2013, Otto et al. 2016, Gallant et al. 2014) or suitability for bee forage (Smart et al. 2016a, Smart et al. 2016b). Landscape-level assessment is a first step to understanding the magnitude or deficit of pollinator resources; however, developing an understanding of plant-pollinator interactions within naturally occurring floral resources and those used in conservation practices is key to the management of bees on CRP and other USDA conservation lands.

Recent and ongoing work by Clint Otto at USGS aims to build a more detailed pollen library data set. Our work complements this assessment of the northern great plains with a more detailed focus on plant-bee interactions in the corn belt and a specific assessment of PHP mixes. PHP has yet to be evaluated and monitored for benefits to honey bees and native bees provided in this specific conservation practice. We present the first review of PHP support for pollinators, focusing on PHP enrollments in WA, MT, IA, and NE seeding in sign-up 39, installed either in 2010 or 2011.

### **Modeling conservation costs and benefits**

Although it is clear that pollinators are intimately linked to crop production and the reproduction of native plants, few attempts have been made to model pollination services at a resolution that is relevant to land management decisions. Models of the potential economic loss or gain associated with pollinator systems have been proposed, often providing large scale estimates with significant standard deviation (see Losey and Vaughan 2006 ; Gallai et al 2009). Suitability mapping or

predicting presence of pollinators has also been attempted, again at a large spatial scale (Gathmann and Tschardtke 2002).

Landscape level mapping of pollinator services and support is a first step, but the data used in these models is insufficient to inform on-the-ground conservation practice. All it can do is answer the question, what do we need? The answer here is more pollinator conservation habitat. These models do not provide resolution on how to approach pollinator conservation from a refined technical perspective. The scale of most pollinator cost-benefit models is too large, or the resolution is not fine enough. In this report we develop a series of site-specific information on pollinator support on PHP lands.

### **Purpose and Deliverables**

This research program represents one of the first quantification and evaluation of PHP seed mixes to honey bees and native bees. Broadly, project deliverables include:

- 1) an initial characterization and assessment of PHP seedings, including specific plant-pollinator interactions;
- 2) quantification of benefits provided to bees by CRP programs, with a particular interest in newly seeded PHP mixes as compared to prior CRP enrollments;
- 3) the development of a descriptive or predictive model outlining PHP and CRP support for managed and wild bees; and
- 4) cost-benefit assessment of PHP compared to prior CRP enrollments, focusing on quantifiable benefits to honey production, noting that economic benefits provided by enhanced native pollinators and other beneficial insects are potentially significant, yet are beyond the scope of this project.

This study was conducted under multiple contracts and extensions with specific goals listed below.

- AG-3151-C-11-0025 and AG-3151-C-14-0001: Monitoring and evaluation of data on PHP and other CRP lands relating to support of honey bees and native bees. Development of either a descriptive or a predictive model support and of costs and benefits provided to honey bees and native bees in CRP systems including PHP and other CRP enrollments.
- AG-3151-P-15-0044: Evaluation of honey bee and native bee preferences for PHP seed mix components. Enhancement on AG-3151-C-14-0001 and AG-3151-C-11-0025.
- AG-3151-P-16-0200 – Quantification of plant-bee interactions and pollinator support expected from specific seed mixes used in PHP. Extension of AG-3151-P-15-0044.

## Methods

### Site selection

Pollinator Partnership worked with FSA staff at the Washington D.C. office to develop a short list of candidate states for study, represented by states with significant pollinator conservation practice enrollments. The initial short list included Texas, Kansas, Illinois, Montana, Nebraska, Iowa, and Washington. Site visits, consultations, and climate factors determined our final selection of Iowa, Montana, Nebraska, and Washington. Within each study state an individual county with significantly high PHP enrollment was selected, including Teton Co., Montana, Lincoln Co., Washington, Woodbury Co., Iowa, and Franklin Co., Nebraska (Figure 1; Appendix 1). Three site types were used in this study, PHP, CRP, and crop sites. Crop sites acted as an overall control for the assessment of any conservation program benefits. PHP, CPR, and crop control sites in each state were selected with the aid of local county FSA staff. Criteria used for selecting each site included placement within a landscape matrix excluding other pollinator conservation practices within a 2 mile radius, and producers willing to participate in the study. Detailed analyses of composition of surrounding landscapes was beyond the scope of the study. While PHP and CRP sites were selected to be as similar as possible in terms of surrounding habitat, it is noted that future quantification and analyses of effects of varying surrounding land matrix on PHP function will be valuable for assessing the benefits of PHP in different land use scenarios. Efforts were made to select PHP sites with varied enrollment sizes in order to assess the impacts of increasing size and pollinator benefits. In our study there were a total of 4 CRP sites and 10 PHP sites. Appendix 2 includes a list of all sites used in the study.

### **Producer identities**

For the purposes of this FSA report, names of producers are anonymous.



Figure 1: Map of county locations of study sites. A) Lincoln Co., Washington, B) Teton Co., Montana, C) Franklin Co., Nebraska, D) Woodbury Co., Iowa. Counties indicated by orange circles.



### Honey bee benefit assessments

The productivity of honey bees foraging within each landscape was measured using weight gained over the season. Automated hive scale units built by HiveSensors LLC were used to collect weight data. Each hive scale unit included three individual scales connected to a data logger that recorded data onto a USB. Scales were powered by solar panels and a back-up battery (Figure 2).



**Figure 2: Hive scale data loggers used for study. Pictured here without hive scales.**

The hive scales were placed on a shipping pallet (as recommended by the manufacturer) to provide a level surface for calibration and measurement (Figure 3). A total of four hives were placed at each site to balance the hive scales units, only three of these hives were actively weighed through the season. The date of scale deployment varied annually according to the schedule of the contracted beekeeper. On average, data were collecting from the late May or early June through the end of September.



**Figure 3: Hive scales with honey bee hives.**

Data loggers were calibrated to collect measures every 10 minutes (developer default) in the first two years of study. In subsequent years data loggers were recalibrated to collect data at 30 minute intervals. Data collected included the date, time, and weight (pounds) of each hive from the day of deployment through to the last day of data collection. Data were logged onto an excel CSV file, overwriting the file at each data collection interval. Field staff collected and copied the data file on a monthly basis to ensure back-up data.

### **Native bee benefit assessments**

Patterns of native bees seen on PHP, CRP, and crop lands were assessed through a variety of methods. A mix of pan trapping and aerial netting was used to outline the bee community at each site, in accordance with Bee Inventory Plot protocols. Pan traps were deployed for one complete sample day every two weeks during the flight season of native bees in the region. Pan traps were deployed at 9:00 am and were collected at 3:00pm. Two 15-minute aerial net transects were carried out during the pan trap collection day, totaling 30 minutes of netting. Figure 4 shows pan traps deployed in the 2014 field season as well as field staff preparing for aerial netting in Nebraska. Specimens collected through pan trapping and aerial netting were pinned and prepared for identification. Identification was carried out to genus by two taxonomists: S. Buchmann and R. Sudan. Pan trap and aerial net collections continued from 2012 through to 2014.



**Figure 4: A) Pan traps used for native bee data collection in Nebraska. B) Field researchers with nets used in aerial netting protocols in Nebraska.**

The productivity of native bees associated with each landscape type was assessed using tube nest occupancy and nest cell provisioning. Five bundles of 30 individual paper tubes inside of a protective cardboard exterior tube were attached to fence posts approximately three feet off the ground, facing a south-eastern direction at each site. Tube nests were assessed bi-weekly throughout the season noting occupancy. At the end of the season, occupied tube nests were collected and prepared for x-ray analysis. Tube nests yielded low data (as is common with this form of data collection) and were only used successfully in the first year of study (2012) in Nebraska, other sites did not yield sufficient results for analysis.

### Plant-insect interaction networks

Plant-pollinator interaction networks were developed using records of bees actively visiting flowering plants in 1 meter by 1 meter quadrat samples. Three randomly placed quadrats were monitored every other week, from spring to late summer, for a period of 5 minutes each (Figure 5). Bee visitors to flowers within the quadrat were noted to lowest possible reliably assessed taxonomic category using visual observations, including *Apis mellifera*, *Anthidium*, *Agapostemon*, *Bombus*, *Ceratina*, *Halictidea*, *Lassioglossum*, *Megachile*, *Melissodes*, *Osmia*, *Xylocopa*, and unknown bee (UB). Both morning and afternoon samples were taken at each site on the sample day to account for daily variability in bee visitation patterns.



Figure 5: Quadrat sample used for plant-insect interaction assessment.

## Data Analysis

### Honey bee support

Data files from each data logger required sorting and cleaning to remove some data errors that were recorded. To reduce the amount of data to a manageable level, a single measure taken between 17:00 and 18:00 on each day was used as the weight as this time of day generally corresponds to the heaviest daily weight hive weight (S. Buchmann, personal communication). Recording errors were removed from the final analysis file. Occasionally, data were not logged in the correct column, likely due to a program error. Additional errors included negative values and unusually high values that often resulted from temporary hive movements during management by beekeepers. There were some occasions during which hive scales were damaged. One scale in Iowa was damaged beyond repair by a grazing cow in 2014. In addition, the connector cables of one scale in Iowa were damaged by field mice. Efforts were made to replace these units; however, they remained damaged through the 2015-2016 sample year. In 2015, one set of hives in Montana was disturbed and damaged by a grizzly bear. The unit was not damaged, but data for that season was lost. As data collection was not consistent between all sites, the start and end date of all usable data was noted and the number of days of data collection were noted.

In some cases, hives did not achieve a weight gain of more than 40 pounds. This is considered a minimum viable weight gain by beekeepers; hives acquiring less than 40 pounds do not survive overwintering periods and are considered losses. Hives gaining less than 40 pounds were removed from data analysis as not to skew the results of viable hives. In addition, all data from the initial sample year (2012) were not included in the analysis as the majority of hives at all sites did not gain 40 pounds. Floral development in 2012 was also minimal as it was the first full growing season and does not represent the full support capacity of the floral mixes.

Hive weight gain was calculated as the maximum weight achieved by the hive during the sample period, minus the starting weight of the hive. In this study, beekeepers did not collect honey produced during the season, allowing weight to accumulate throughout the season. The absolute weight gain for each individual hive and the mean gain for each site were used in subsequent statistical analysis. Mean gains across each landscape type (PHP, CRP, and crop) across all states and between sites within each state were compared using ANOVA in R. Single and multivariate regression was used to compare mean gains in PHP and CRP to the richness of the seed mix planted, size of seeded areas, and size of total enrollment (PHP only).

### Tube nest assessment – x-ray

Occupied nest tubes from each site were fixed to cardboard using clear tape, in preparation for x-ray analysis. X-rays at 50kv for 30 seconds were taken of the total occupied tubes from each landscape type to allow an assessment of larval development and cell number in each tube. Digital files of the x-rays were provided and used for assessment. Occupancy numbers and nest cells provisioned were compared between the three landscape types using multiple non-parametric test to account for the small sample size.

### Native pollinator support

Mean richness (genus-level) and abundance of native pollinators at each landscape type (PHP, CRP, and crop) across all states and between sites within each state were compared using ANOVA. Single and multivariate regression were used to compare mean richness and abundance in PHP and CRP to the richness of the seed mix planted, size of seeded areas, and size of total enrollment (PHP only).

Descriptive assessments of the native bee community on each landscape type and in each state were provided, noting the presence and absence of key pollinator groups, functional guilds, and other relevant ecological characteristics using multiple sources.

### Plant-Pollinator networks

Plant-Pollinator interactions were assessed in R Bipartite Package (Dormann et al. 2008) for each site and collective network for CRP and PHP. Network level and species level analyses were conducted for each site and for each treatment (CRP or PHP) (Dormann et al. 2009). We report on some of the indices generated by R Bipartite for CRP and PHP sites. Plants were sorted based on number of bee visits, number of bee visual ID groups, and number of honey bee and native bee visits. Using a quantitative score for rank in each of these four sorting categories, plants were given a value of one to five (five being the highest value in that category, one being the lowest). Scores were added to create a 'Plant Value Index' to bees. Sites were the same in Iowa in 2015 and 2016 and data from the same sites were pooled. There were a total of six PHP and three CRP sites in plant-pollinator interaction analyses.

### Cost-Effectiveness Analyses

Cost analyses including enrollment costs, honey production, and bee community enhancement was developed to compare CRP and PHP landscapes (Table 1). Benefits are presented in monetary and non-monetary values, when possible, for comparison. However, we note that the evaluation of native bee benefits is best considered in a non-monetary context as markets for biodiversity are not well-developed. FSA provides a 50% cost share to seed mix costs, with the producer providing the remaining 50%. We present costs as the initial total amount for seed and SIP (Table 2).

**Table 1. Cost and partial benefits associated with CRP and PHP enrollments (not including other cost such as installation).**

Enrollment Type	Costs	Benefits
CRP	Seed mix costs	Honey bee weight gain (pound of honey) Native bee abundance Native bee richness
PHP	Seed mix costs SIP per acre enrolled	Honey bee weight gain (pound of honey) Native bee abundance Native bee richness

The cost and benefit enhancement of PHP over CRP can be visualized as:

$$PHP_{benefit\ enhancement} = (\sum CRP_{cost} - \sum PHP_{cost}) + (\sum PHP_{benefits} - \sum CRP_{benefits})$$

A negative value for PHP benefit enhancement indicates that PHP costs are greater than the sum of the benefits, whereas a positive value indicates that PHP costs are made up for in benefits. Because monetary and non-monetary values are included in the above equation, it is a conceptual model.

There are many potential benefits of PHP. Some of these include increased honey production by managed honey bees, healthier honey bee colonies (and lower colony loss or treatment requirements), increased abundance and diversity of native bees and other pollinators, improved wildlife habitat and increased abundance and diversity of wildlife, and greater service provision on nearby cropped lands (such as enhanced pollination and pest control). For many of these potential benefits of PHP over CRP, it is not possible to directly quantify benefits in terms of dollars.

Therefore, the cost-benefit analysis uses only increased honey production as the monetized benefit, acknowledging that there may be many other economic benefits that are difficult to quantify and not included in this analysis. This makes the cost-benefit analysis an underestimate of the possible economic benefits of PHP enrollment. However, it provides a starting point to assess whether some of the increased costs of PHP over CRP are offset by direct economic benefit.

Similarly, benefits of PHP over CRP to native bee abundance and diversity were analyzed in terms of cost differential of the practices in relation to proportional enhancement of bee communities. It is noted that these are only partial benefits and many other potential benefits are not included in this cost-effectiveness analysis, again, making this an underestimate of potential benefits.

**Table 2. Seeding costs per site for CRP and PHP**

Site	State	Type	Cost of seeding	Size (acres)	Pollinator Size(acres)	No. of Pollinator plants seeded	CRP cost/acre	PHP cost/acre
CRP-1	IA	CRP	\$ 1,115.39	13.6	na	2	\$ 82.01	
CRP-2	MT	CRP	\$ 5,112.93	202	na	2	\$ 25.31	
CRP-3	NE	CRP	\$ 201.12	15	na	1	\$ 13.41	
CRP-4	WA	CRP	\$ 808.29	264.6	na	2	\$ 3.05	
PHP-1	IA	PHP	\$ 1,683.00	32	10.2	22		\$ 165.00
PHP-2	IA	PHP	\$ 621.16	25	2.5	12		\$ 248.46
PHP-3	IA	PHP	\$ 663.40	42.2	3.1	21		\$ 214.00
PHP-4	MT	PHP	\$ 282.10	9.1	0.9	6		\$ 313.44
PHP-5	MT	PHP	\$ 154.80	15.9	1.8	6		\$ 86.00
PHP-6	NE	PHP	\$ 224.50	17.5	1.9	36		\$ 118.16
PHP-7	NE	PHP	\$ 348.47	37.6	5.3	33		\$ 65.75
PHP-8	NE	PHP	\$ 224.50	44.3	4.9	25		\$ 45.82
PHP-9	WA	PHP	\$ 243.96	25.91	4	10		\$ 60.99
PHP-10	WA	PHP	\$ 359.80	28.2	2.8	10		\$ 128.50

## Results

Results for honey bee productivity and native bee richness and abundance on are based on a series of analyses, grouping treatments by regions and overall by treatment type. Individual patterns and trends for each state and for each site also are presented and have been reported to County FSA offices. Bee-plant interactions are based on aggregated data for plant and pollinator species at each individual site and for each enrollment category (CRP and PHP).

### Honey bee productivity

Over the four years of monitoring from 2013 to 2016 honey bees on PHP landscapes gained a mean of  $122.09 \pm 10.25$  pounds each season, while CRP hives gained an average of  $74.54 \pm 9.03$  pounds and crop hives gained an average of  $74.02 \pm 4.72$  pounds (Figure 6). Honey bee colonies placed on PHP landscapes gained significantly more weight when compared to CRP lands ( $p=0.010$ ) and croplands ( $p=0.003$ ). Weight gains on CRP lands were not significantly different than those seen crop landscapes ( $p=0.859$ ).

State level patterns displayed a consistent trend, with PHP hives gaining more weight, on average, compared to CRP and crop sites (Figure 7, 8, and 9). Significantly higher gains across all sites were driven by trends seen in Iowa PHP sites, which on average gained more weight than hives in Nebraska and Montana (Figure 10). Generally, hive weight gain was highest in Iowa on all landscape types, when compared to Nebraska and Montana (Figure 10).

**Multivariate and univariate regressions did not identify any significant relationships between PHP enrollment size, total CRP size, or seed mix richness and hive weight gain ( $p>0.05$ ) (Table 3). and**

Figure 12 display the lack of linear relationship between the two key variables of interest, enrollment size and seed mix richness. Again, the categorical variable of enrollment in PHP was the only significant explanatory variable for hive weight gain.

**Table 3. Linear model out for variables correlating in honey bee support.**

Honey bee weight gains	F	DF	p-value	r <sup>2</sup>	r <sup>2</sup> - adjusted
	1.04	3,16	0.40	0.1634	0.0065
Variable		Estimate	Std. Error	t-value	p-value
PHP Size		5.41	4.78	1.13	0.274
Plant Richness		-1.46	1.15	-1.27	0.221
CRP size		0.46	0.99	0.46	0.652

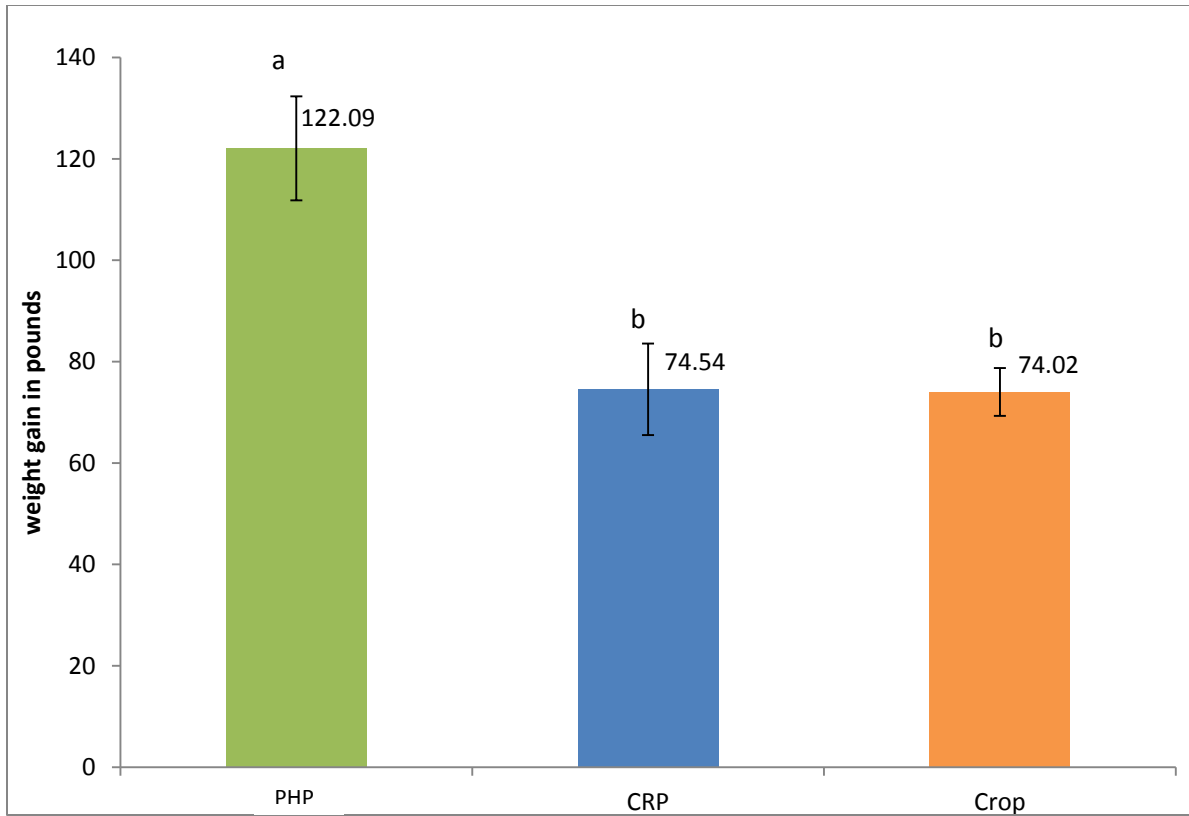


Figure 6. Comparative honey bee productivity in PHP, CRP, and crop fields from 2013-2016. Similar lower-case letters denote means that are not significantly different.

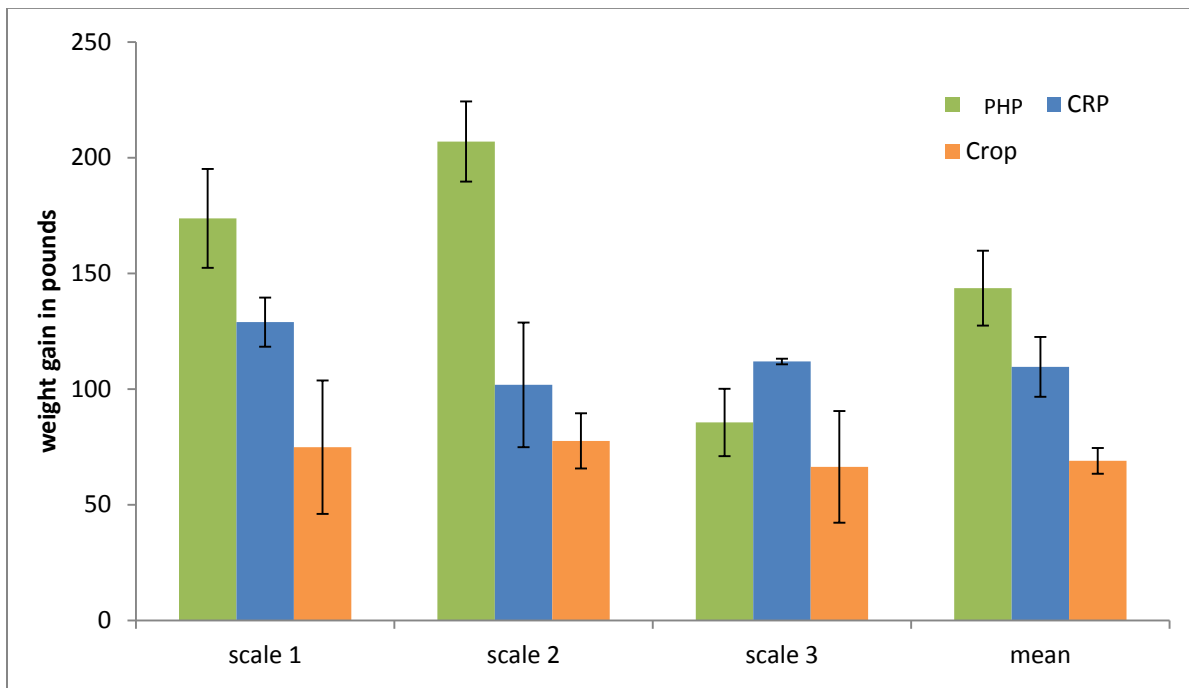


Figure 7. Hive weight gains across PHP, CRP, and crop in Iowa from 2013-2016. Data from each individual scale and the mean for all scales at the site are reported.



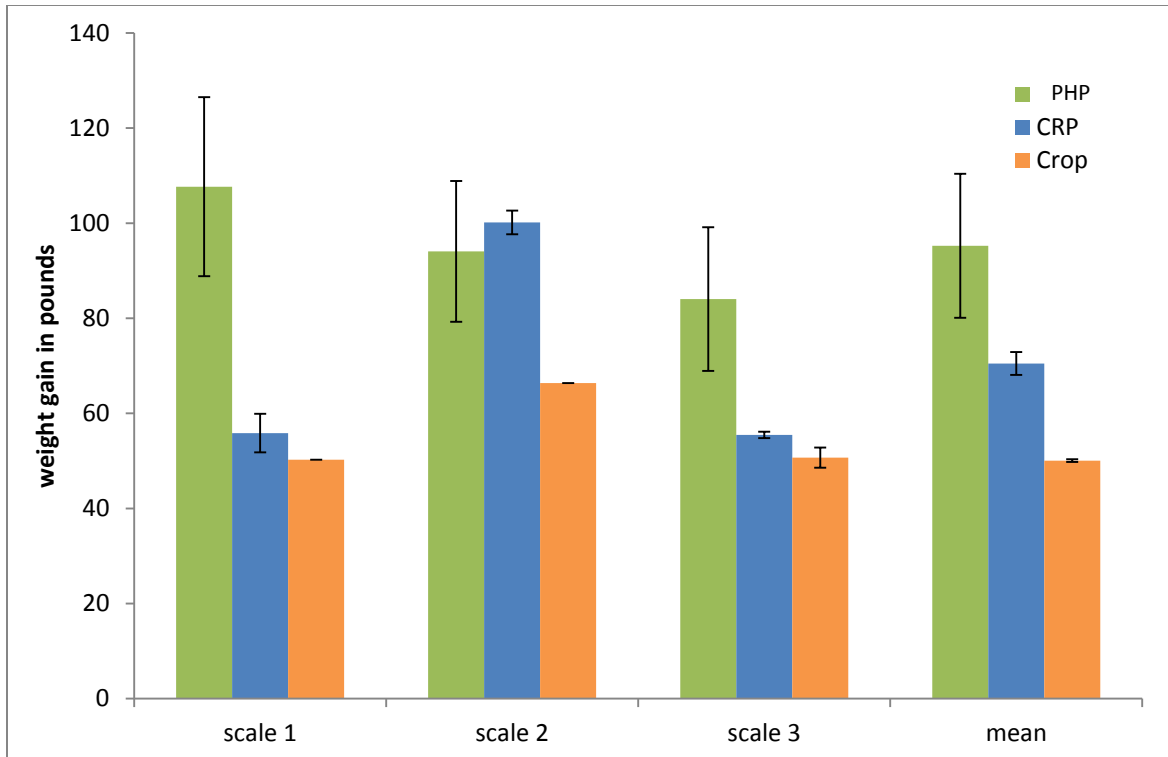


Figure 8. Hive weight gains across PHP, CRP, and crop in Nebraska from 2013-2016. Data from each individual scale and the mean for all scales at the site are reported.

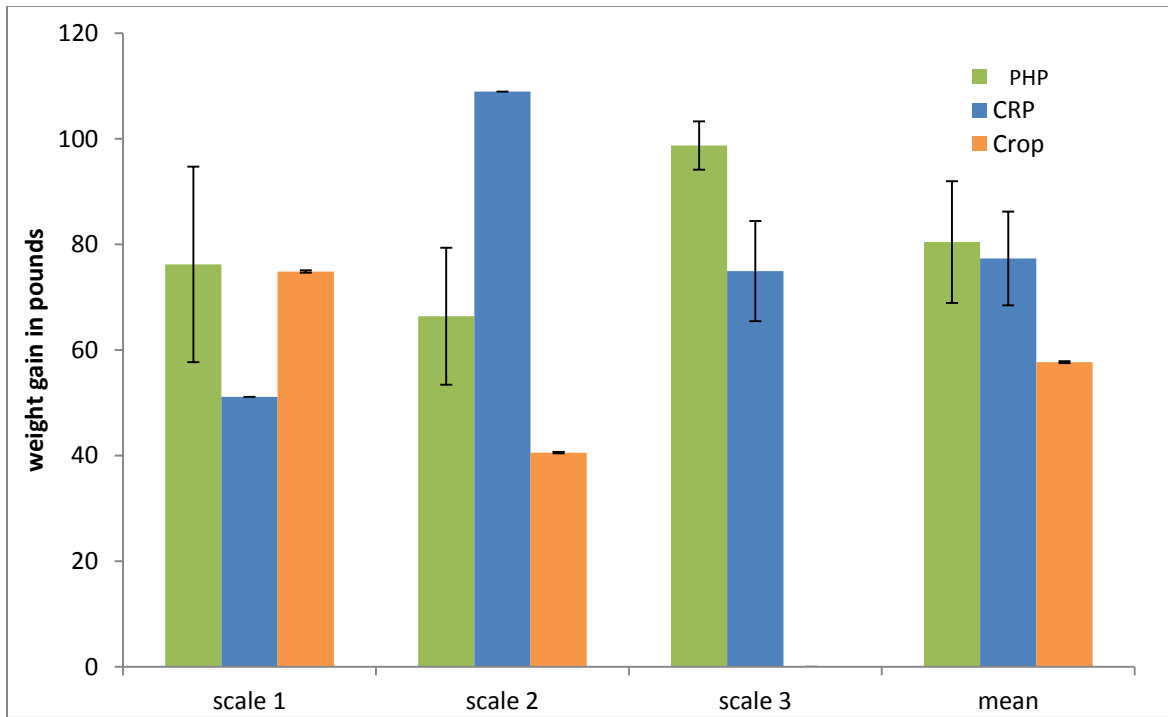


Figure 9. Hive weight gains across PHP, CRP, and crop in Montana from 2013-2016. Data from each individual scale and the mean for all scales at the site are reported.

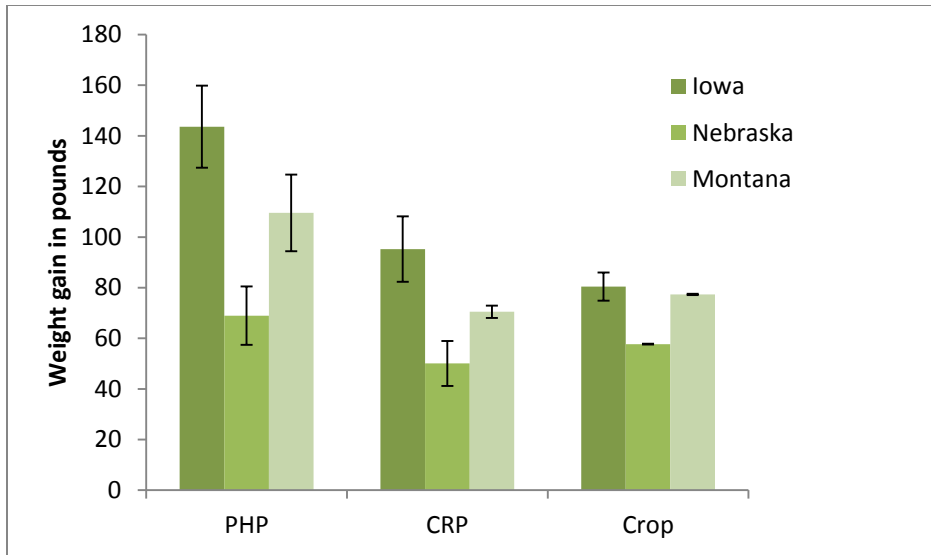


Figure 10. Honey bee hive weight gain at PHP, CRP, and Crop sites in each state.

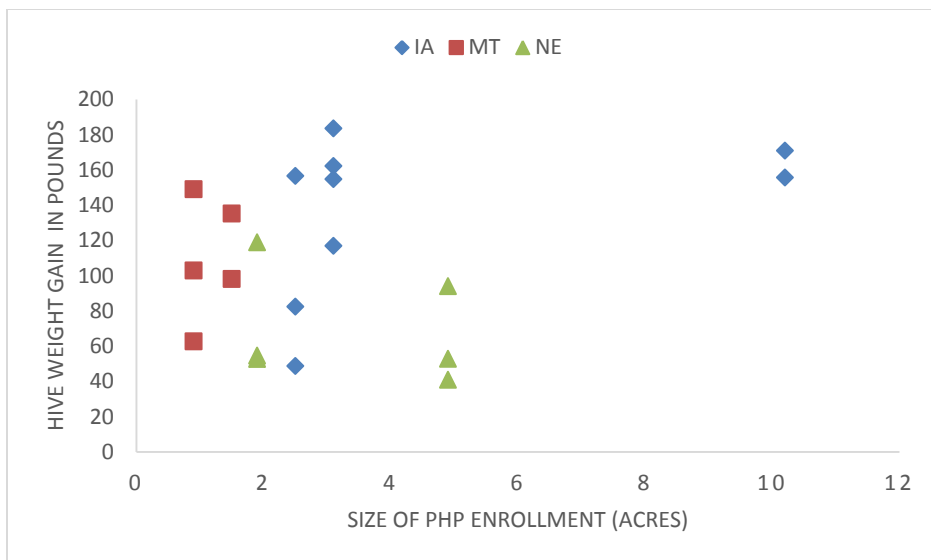


Figure 11. Hive weight gain as a function of PHP enrollment size, coded by state; no significant relationship,  $p=0.27439$ ;  $R^2=0.0781$ .

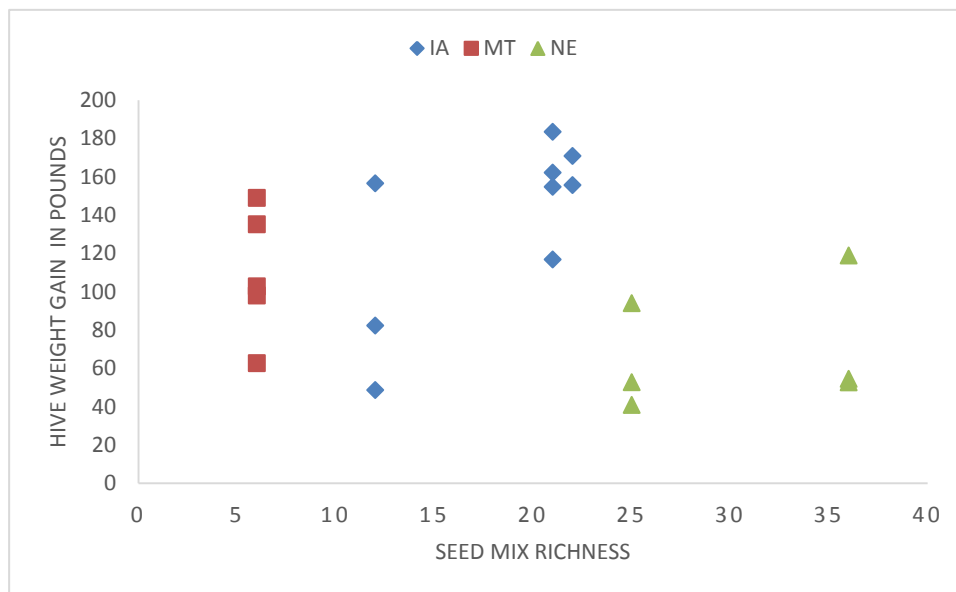
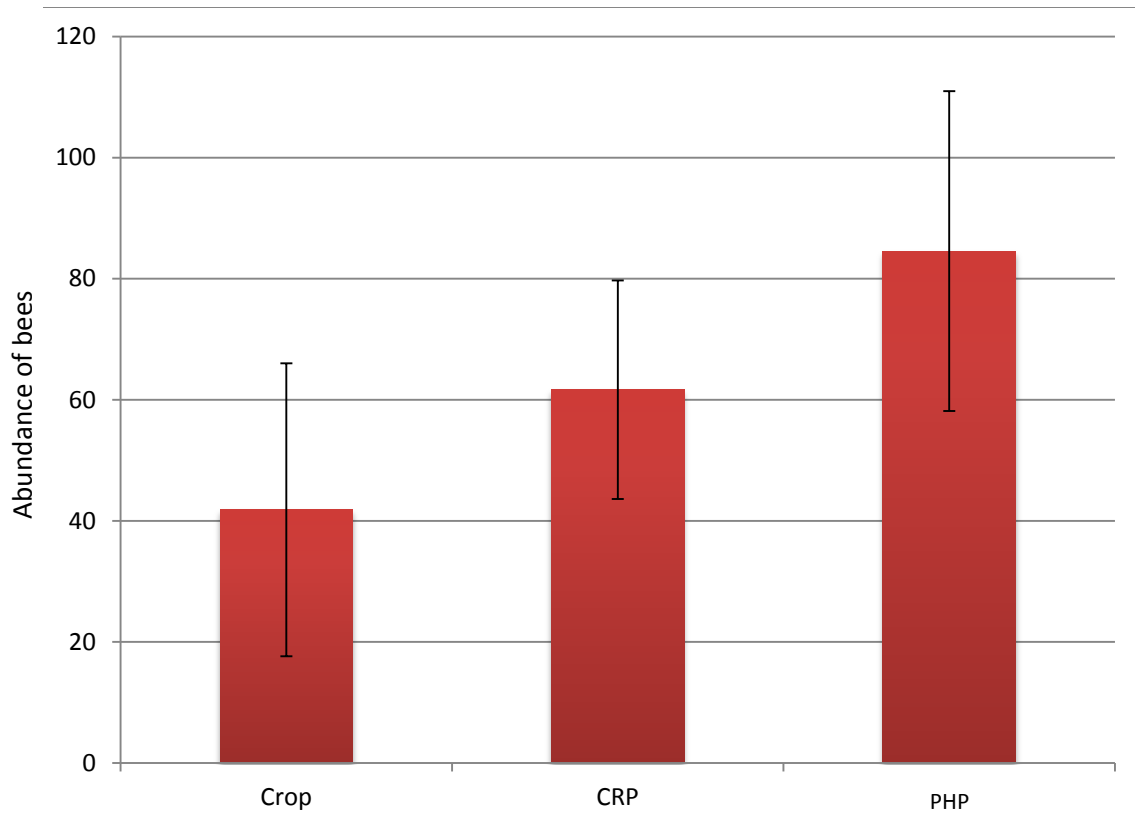


Figure 12. Hive weight gain as a function of PHP seed mix richness, coded by state; no significant relationship,  $p=0.22103$ ;  $R^2=0.0313$ .

### Native bee support

The abundance of native bees collected using pan traps and aerial nets showed a significant difference among land types, with crop landscapes having the lowest ( $41.83 \pm 24.19$ ), CRP having higher numbers ( $61.67 \pm 18.05$ ), and PHP having the highest ( $84.57 \pm 26.14$ ) (Figure 13). Although PHP abundance was 27% higher, this value was not statistically significant due to small sample size ( $p=0.483$ ). Richness displayed a similar increasing trend with crop lands having the lowest ( $6.17 \pm 2.46$ ), CRP being higher ( $7.33 \pm 1.52$ ) and PHP being the highest ( $12.36 \pm 2.25$ ). Although richness on PHP was nearly 40% higher than CRP richness this value was not statistically significant due to small sample size ( $p=0.081$ ) (Figure 14). Regression tests of site characteristics, including overall site size, size of PHP seeding, and PHP seed mix richness showed a positive relationship between abundance and richness of bees (at the taxon and generic level) in relation to PHP size (Table 4). This relationship, while significant, had low to moderate fit with and  $r^2$  of less than 50% in all cases.

These results suggest that for every acre increase in PHP an additional 14.9 native bee visits can be expected (Table 4) using these sample methods. Similarly, for every acre increase in PHP an additional 1.02 taxa can be expected (Table 4) with these sample methods. Measured bee abundance and richness numbers would vary depending on sampling techniques and intensity. However, slopes of lines (rate of change among variables) should be similar among sampling techniques and intensity that adequately sample the population.



**Figure 13.** Abundance patterns of native bees on each landscape type across all states (NE, IA, MT, WA). An increasing trend is seen from crop, to CRP, to PHP, however this increase between land-use types is not significant.

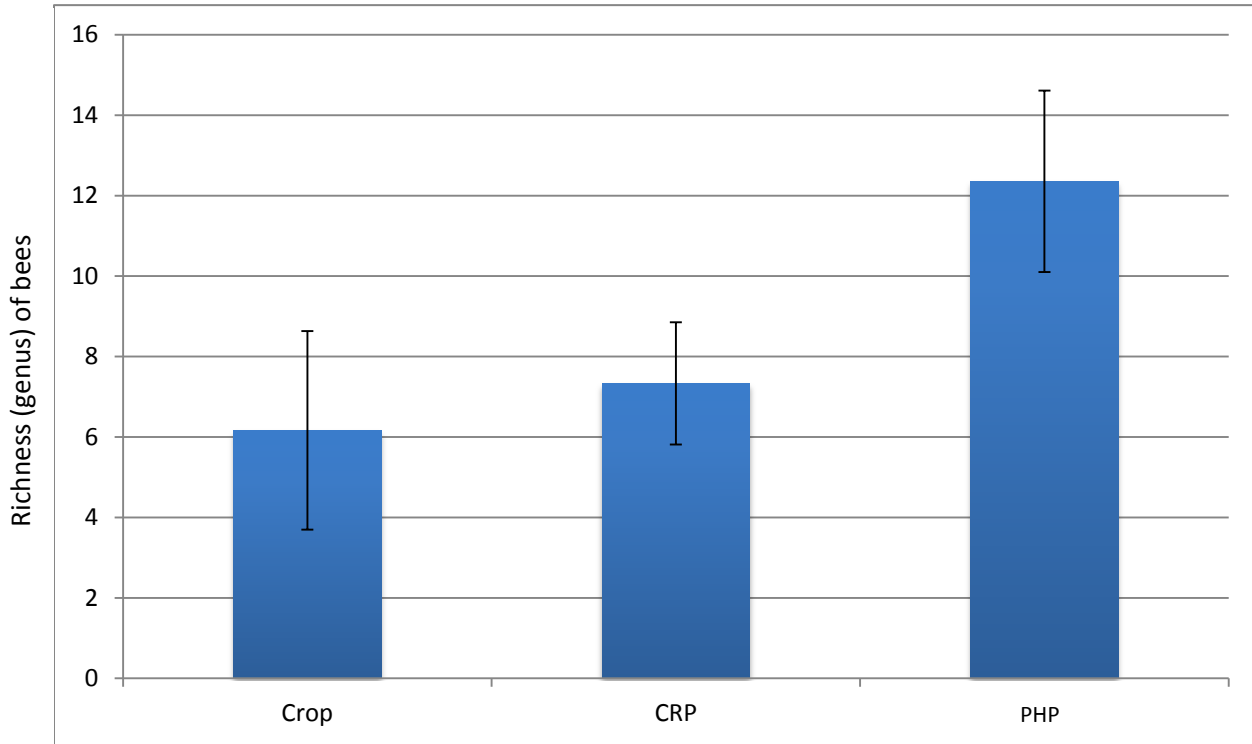


Figure 14. Richness (at the level of genus) patterns of native bees on each landscape type across all states (NE, IA, MT, WA). An increasing trend is seen from crop, to CRP, to PHP, however this increase between land use types is not significant.

**Table 4 Linear regression models for native bee responses to PHP variables.**

Native bee abundance	F	DF	p-value	r <sup>2</sup>	r <sup>2</sup> - adjusted
	5.18	3,16	0.01	0.4167	0.3843
Variable	Estimate	Std. Error	t-value	p-value	
PHP Size	14.88	3.79	3.93	<b>0.001</b>	
Plant Richness	1.61	1.51	1.06	0.303	
CRP size	0.19	0.20	0.95	0.256	
Native bee taxon richness	F	DF	p-value	r <sup>2</sup>	r <sup>2</sup> - adjusted
	3.11	3,16	0.06	0.3687	0.2503
Variable	Estimate	Std. Error	t-value	p-value	
PHP Size	1.02	0.38	2.72	<b>0.015</b>	
Plant Richness	0.03	0.15	0.18	0.857	
CRP size	0.01	0.02	0.41	0.686	
Native bee genus richness	F	DF	p-value	r <sup>2</sup>	r <sup>2</sup> - adjusted
	4.86	3,16	0.01	0.4766	0.3784
Variable	Estimate	Std. Error	t-value	p-value	
PHP Size	0.45	0.17	2.72	<b>0.015</b>	
Plant Richness	0.09	0.07	1.42	0.176	
CRP size	0.01	0.01	0.68	0.505	
Native bee family richness	F	DF	p-value	r <sup>2</sup>	r <sup>2</sup> - adjusted
	2.78	3,16	0.07	0.3427	0.2195
Variable	Estimate	Std. Error	t-value	p-value	
PHP Size	0.09	0.05	1.76	0.090	
Plant Richness	0.01	0.02	0.27	0.793	
CRP size	0.00	0.00	1.86	0.082	

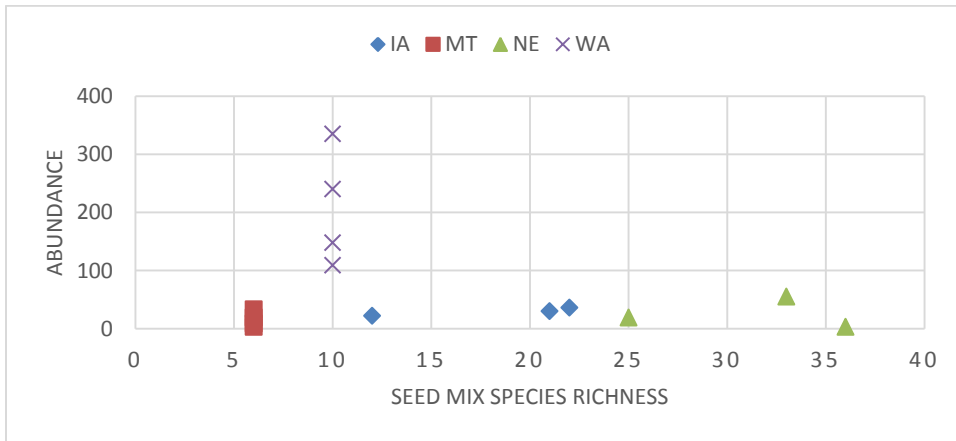
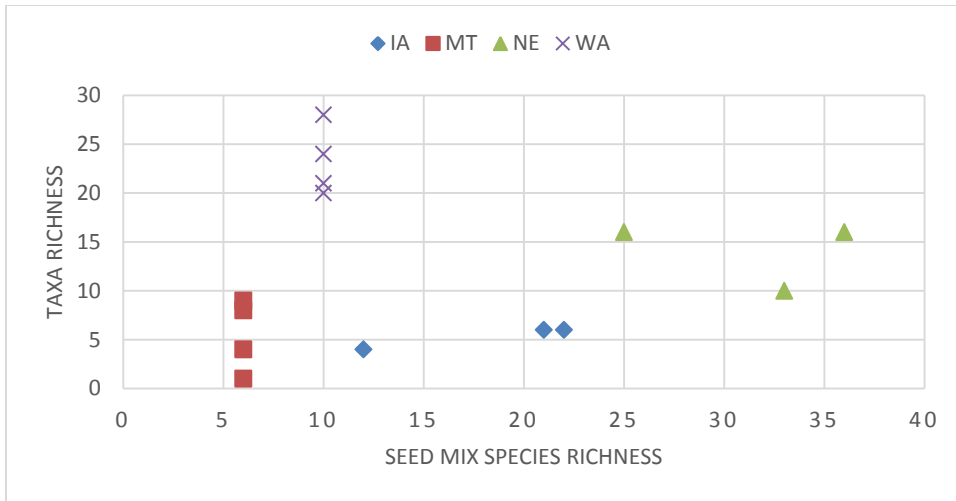
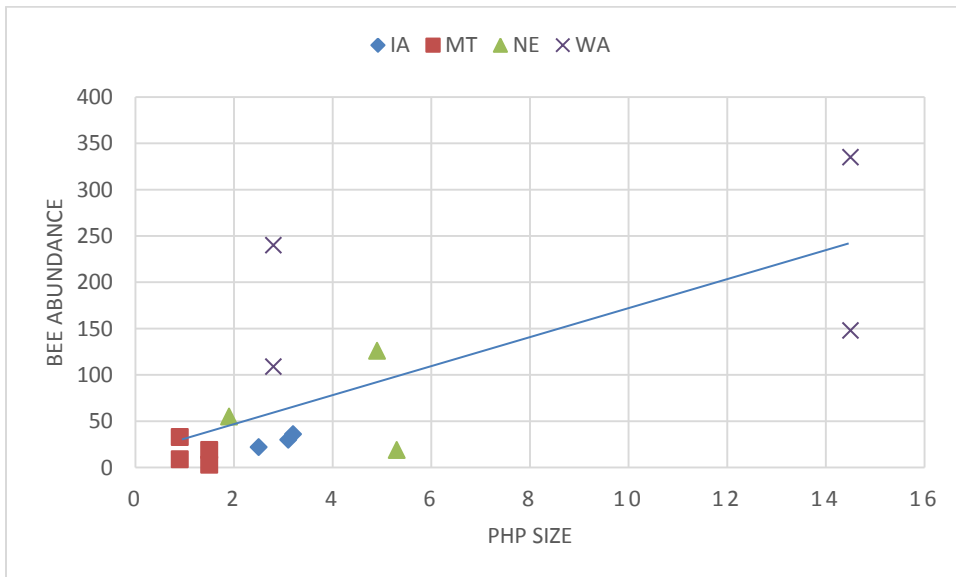


Figure 15. Relationship between native bee abundance and richness and the seed mix richness of PHP mixes, coded by state. Trend lines are not shown if the relationship was not significant at  $p < 0.10$



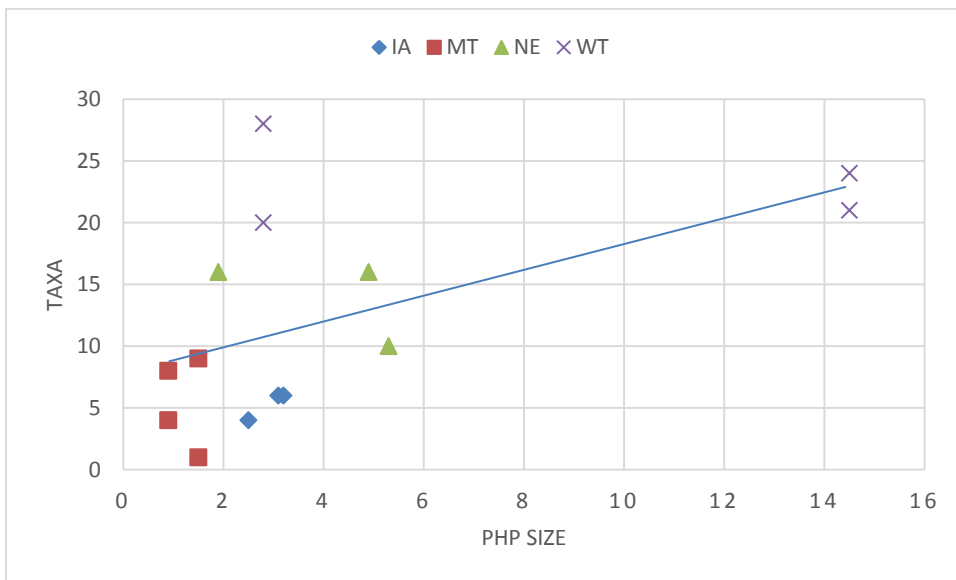
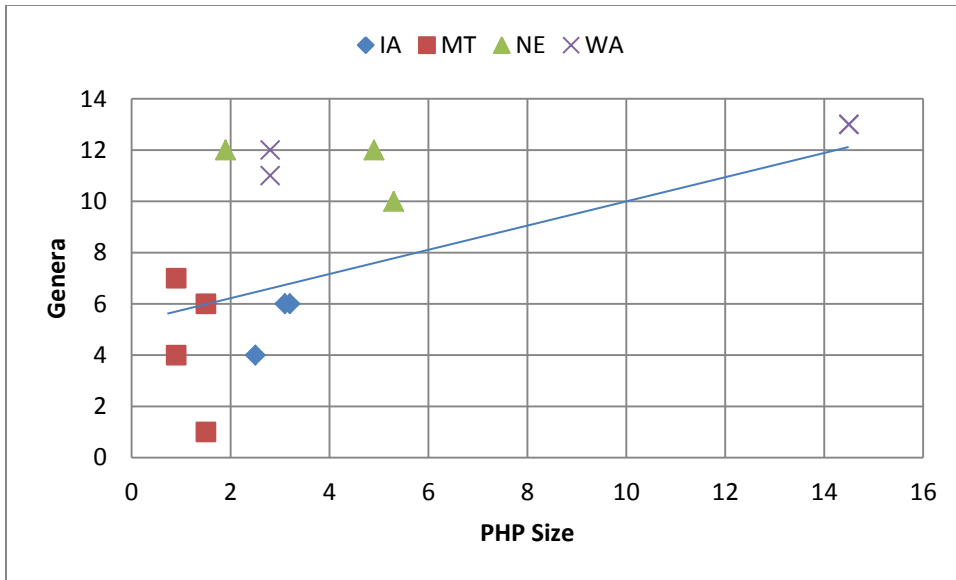


Figure 16. Relationship between recorded bee abundance and richness (taxon and generic) as a function of increasing PHP seeded area size, coded by state. Trend lines are not shown if the relationship was not significant at  $p < 0.10$



## Nesting

Nest site occupancy yielded measurable results in only one state, Nebraska. Over the course of the 2012 season an increasing occupancy trend was seen, with PHP having the overall highest nest occupancy, however, these differences were not significant (Figure 17-18;  $p=0.270$ ). X-rays conducted on each of the capped nest tubes for each landscape type provided an assessment of nest cell provisioning and successful larval development. These data are insufficient in number for statistical assessment, but provide a quantitative assessment indicating that PHP landscapes had higher nest use and successful larval provisioning compared to CRP and crop landscapes (Table 5). X-ray images for tube nests are presented here for interest (Figure 19).

Table 5. Nest tube occupancy statistics from Nebraska, 2012

	PHP	CRP	Crop
Number of capped nest tubes	39	24	12
Number of occupied nest	25	11	3
Total larval number	148	65	10
Mean larval number per nest	4.9	6	5

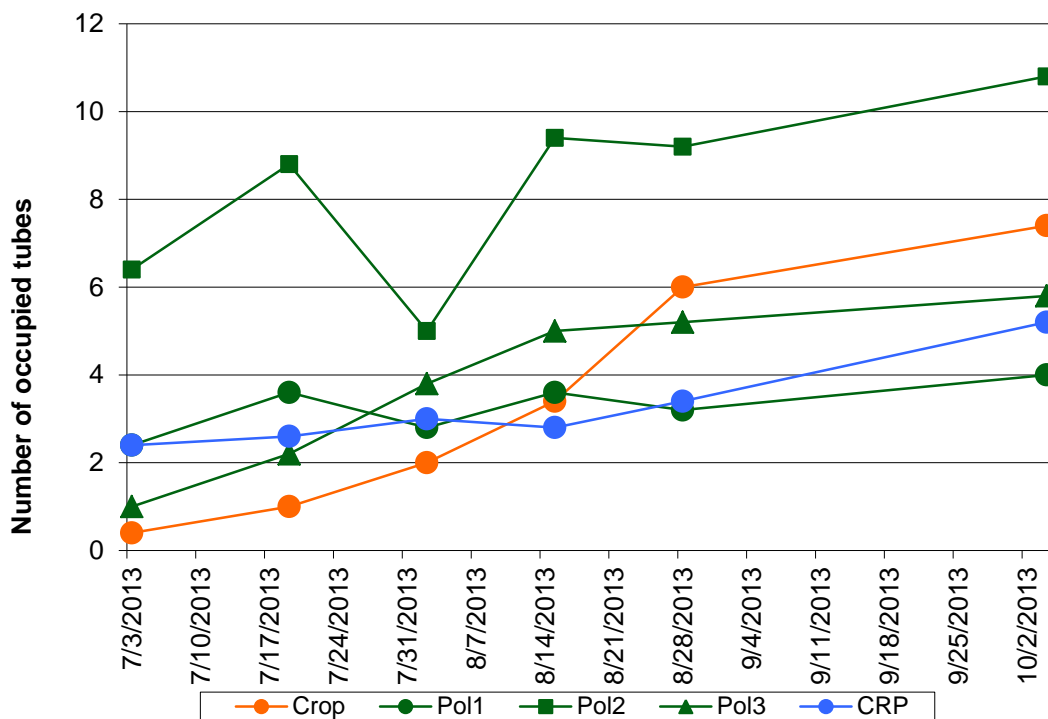


Figure 17. Patterns of nesting occupancy throughout the 2012 season in Nebraska.

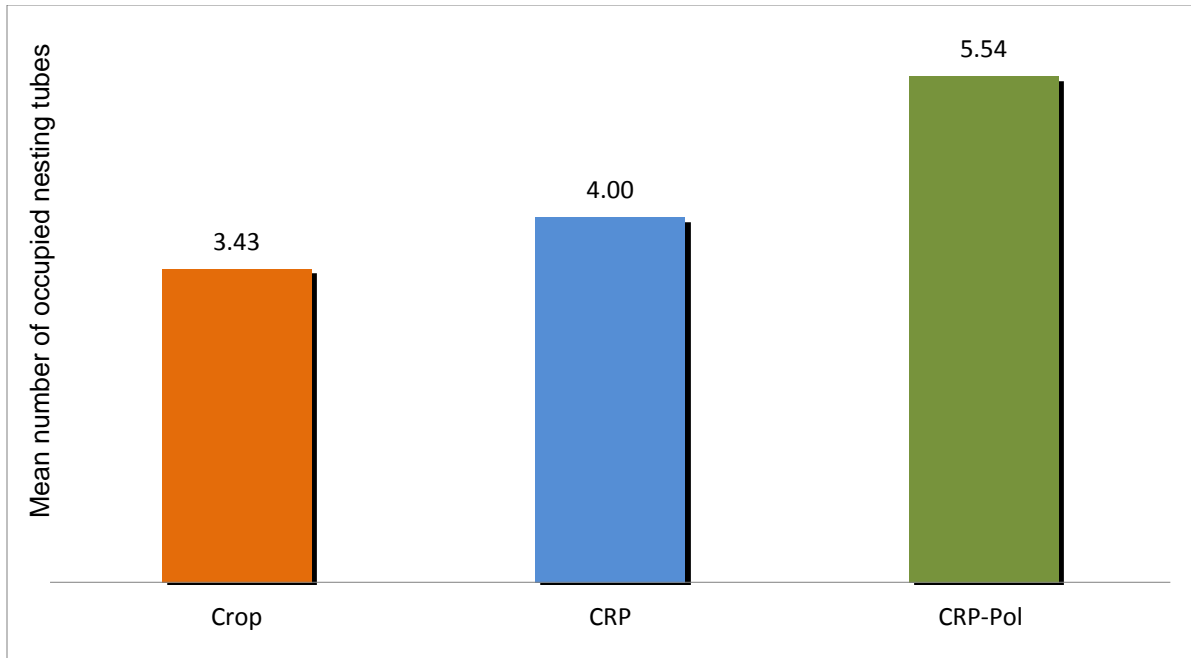


Figure 18. Mean number of nesting tubes occupied (capped) in the 2012 season. Sample size insufficient for statistical analysis.

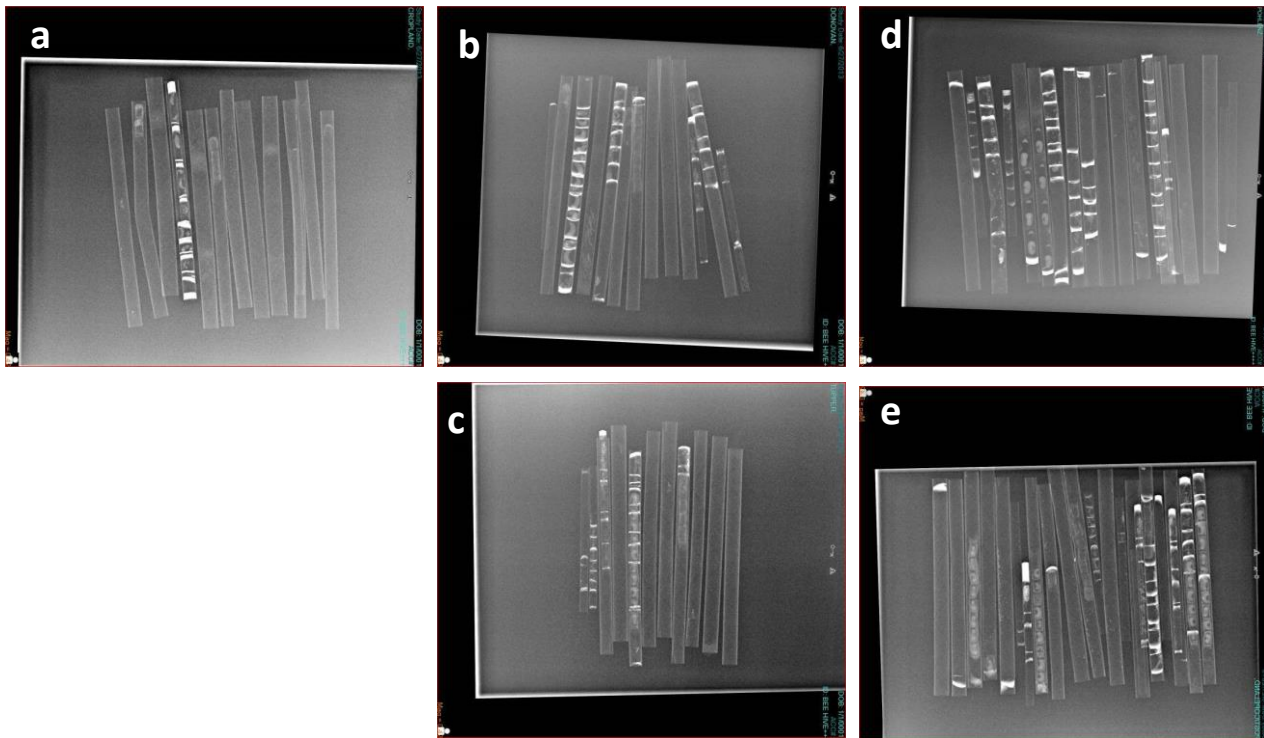


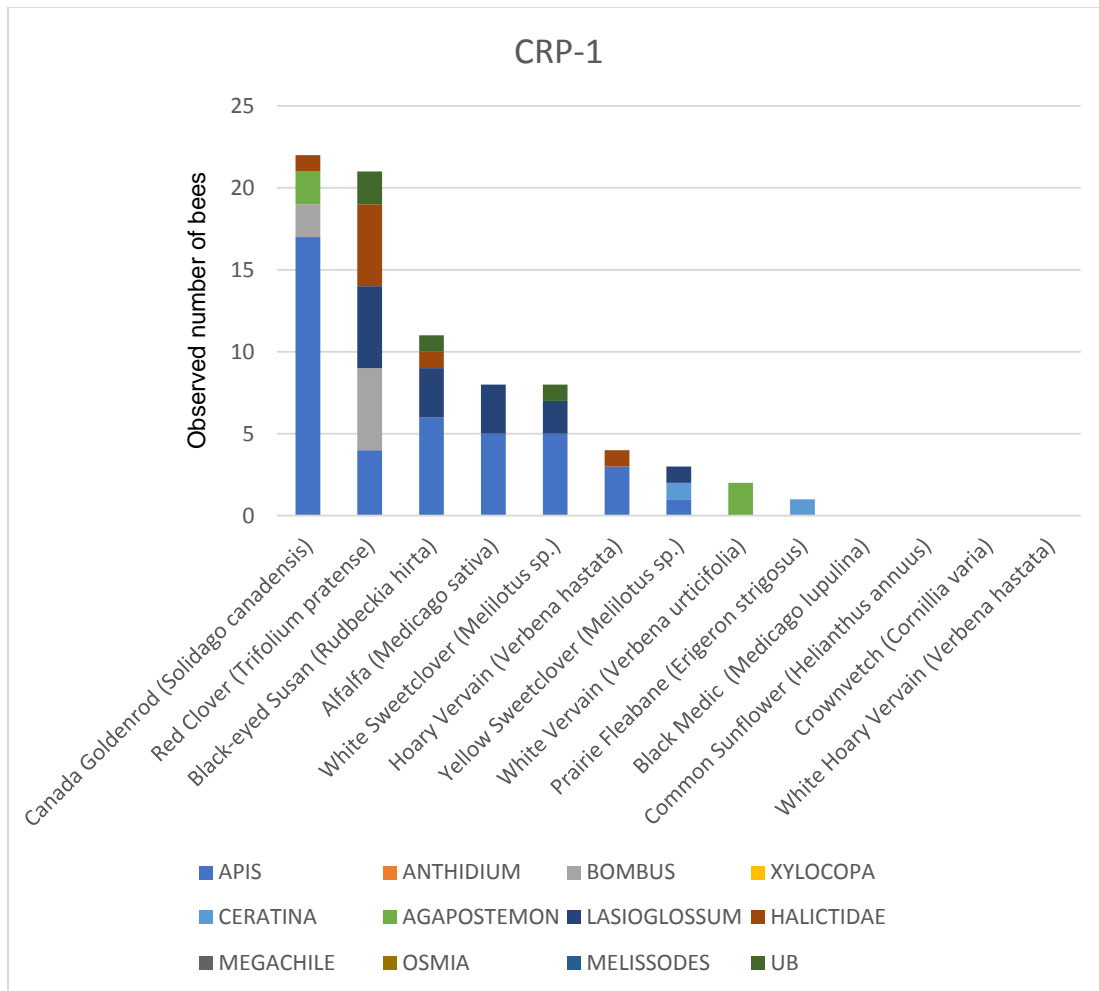
Figure 19. X-rays of capped tube nests collected from all sites in Nebraska, 2012. Crop (a), CRP (b, c), and PHP (d,e).

### Plant pollinator interactions

A total of 683 plant-bee interactions were recorded at PHP and CRP sites, on 51 different plant species. Approximately half the interactions observed on flowers involved native bees (334 of 683) and the rest were honey bees. Interactions on PHP sites involved 286 native bees and 247 honey bees, on 47 different plant species. At CRP sites there were 48 native bees and 108 honey bees observed on 15 different plant species (note that totals are cumulative from six PHP and two CRP sites, therefore the sampling effort at CRP sites was one third the effort of the PHP sites). PHP sites had a higher proportion of native to honey bees than CRP sites. However, we did not perform analyses on these data due to the difference in sampling effort and the relatively small number of CRP sites in the plant-pollinator interaction observations.

There was a high amount of variation in plant use based on species. Some plant species were used by a high variety and/or abundance of bees, other plants (either planted in seed mixes or volunteers) had no recorded interactions with bees during the observations in this study. While this does not mean that these plants are not used by bees, there is a likelihood that they provide less overall benefit. We present the information on interactions in bar graph and bi-partite interaction networks, and provide a quantitative analyses of plant value to pollinator support with recommendations of plants to include in seed mixes, and those that likely provide less support in terms of number of bees or number of taxa supported. It is noted that higher plant diversity can result in more types of bees supported, but cost increases with greater diversity seeded likely will have decreasing returns in terms of numbers of bees and taxon increases.

The following graphs and Bi-Partite diagrams are two ways to visualize bee-plant interactions. The graphs show the relative numbers of the different groups of bees on each flower type observed at the site during the sample period. The Bi-Partite diagrams show two-way strength of association between each bee group observed and each plant species that was involved in an interaction. The width of the black bars indicate relative number of interactions that that bee group or plant species was involved in (at that site), and the lighter bars joining the two trophic levels indicate relative individual magnitude of association between those two groups.



**Figure 20. Abundance of groups of bees on plant species at site CRP-1. Plant species with no bar were present in the observation area (and in flower) but had no bees observed on them during the sampling periods.**

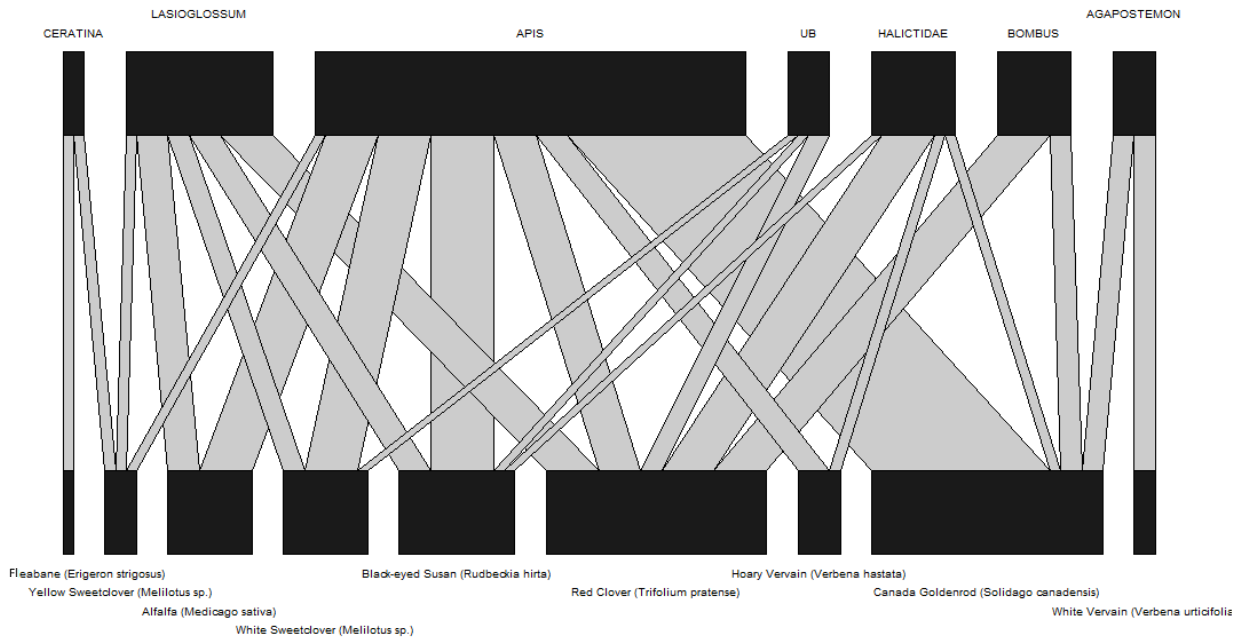


Figure 21. Bipartite interaction network between bee groups and plant species at site CRP-1.

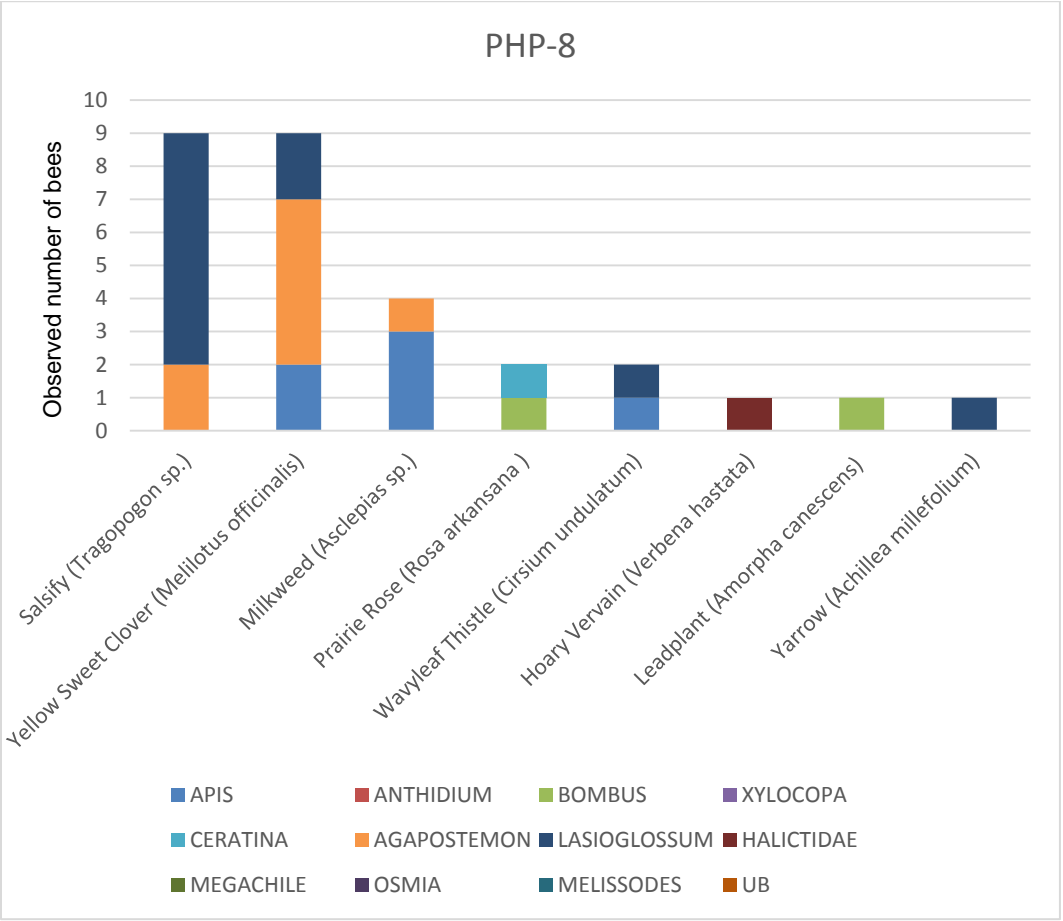


Figure 22. Abundance of groups of bees on plant species at site PHP-8.

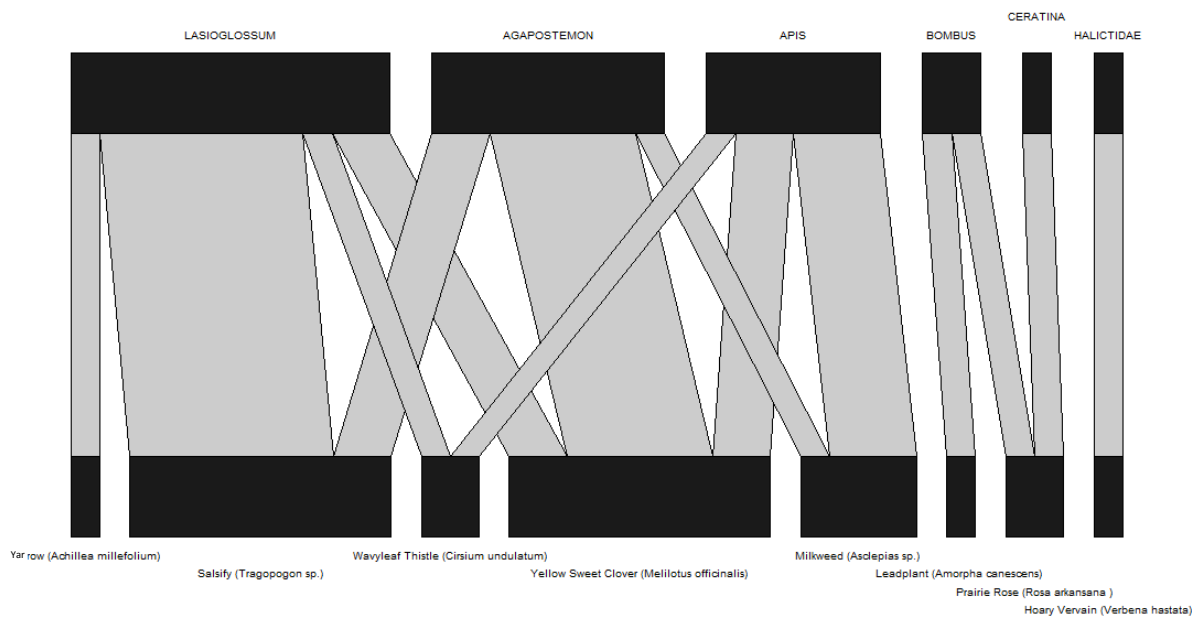


Figure 23. Bipartite interaction network between bee groups and plant species at site PHP-8.

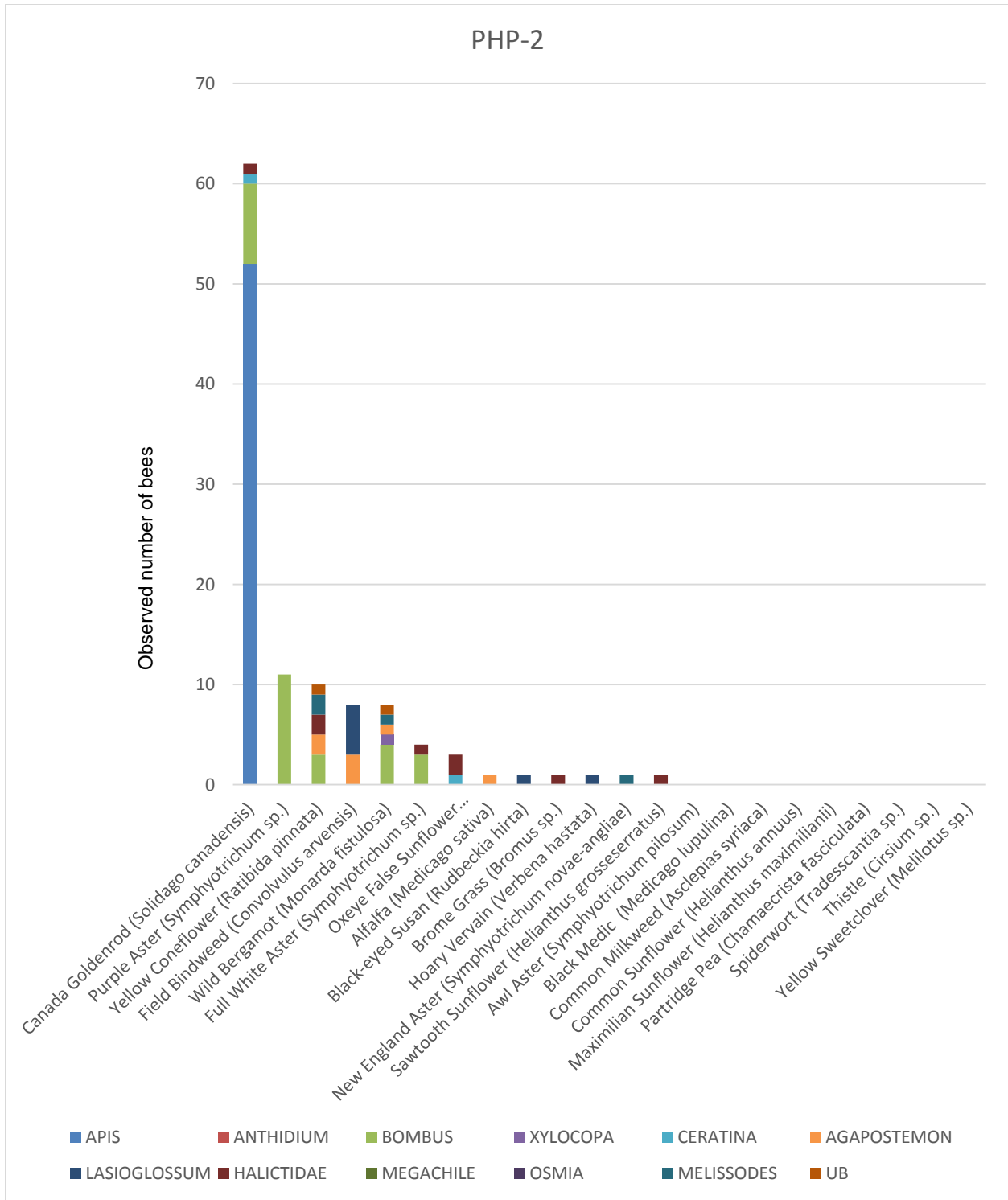


Figure 24. Abundance of groups of bees on plant species at site PHP-2. Plant species with no bar were present in the observation area (and in flower) but had no bees observed on them during the sampling periods.



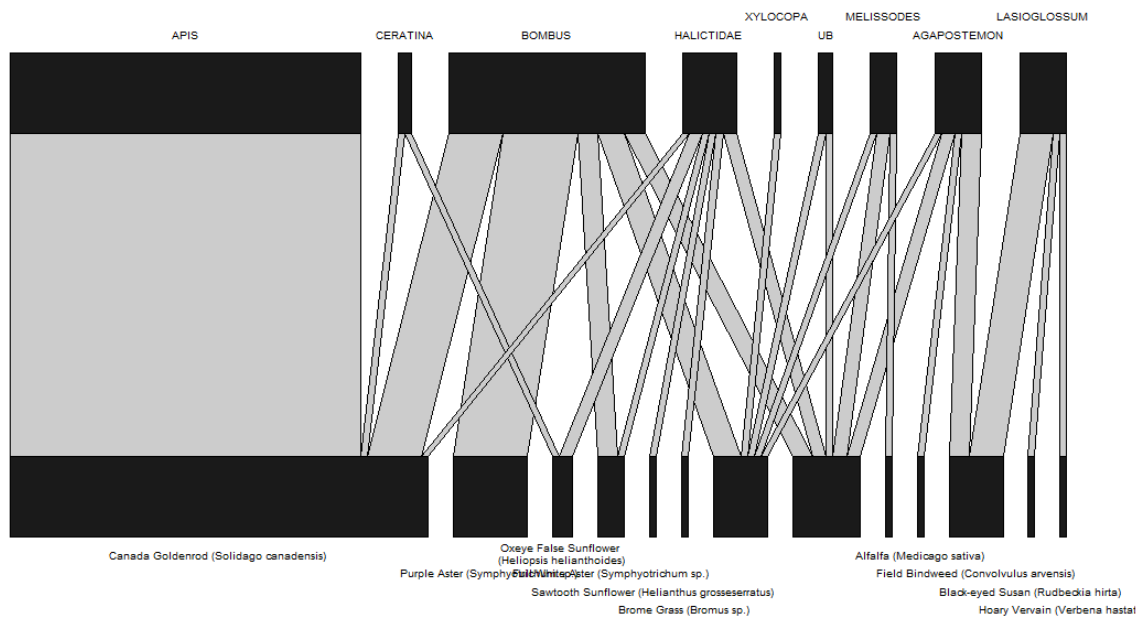
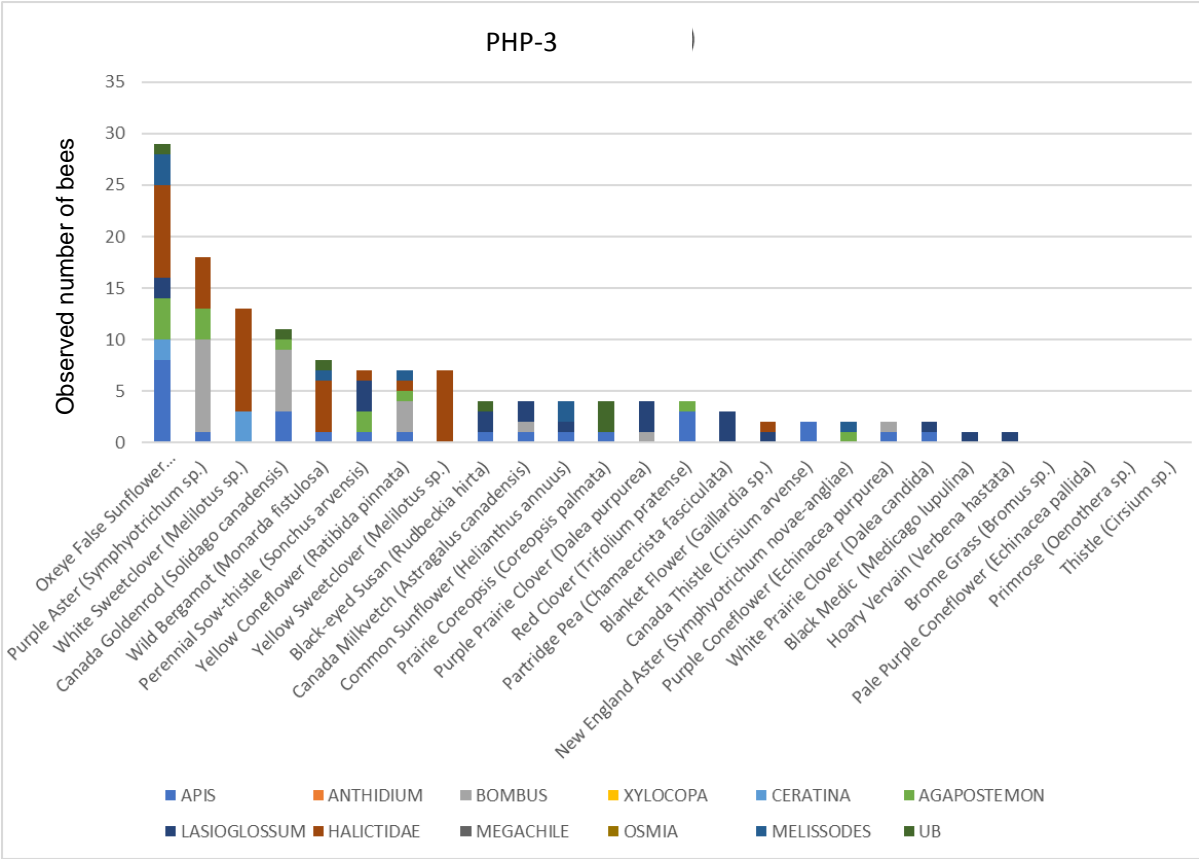


Figure 25. Bipartite interaction network between bee groups and plant species at site PHP-2.



**Figure 26. Abundance of groups of bees on plant species at site PHP-3. Plant species with no bar were present in the observation area (and in flower) but had no bees observed on them during the sampling periods.**

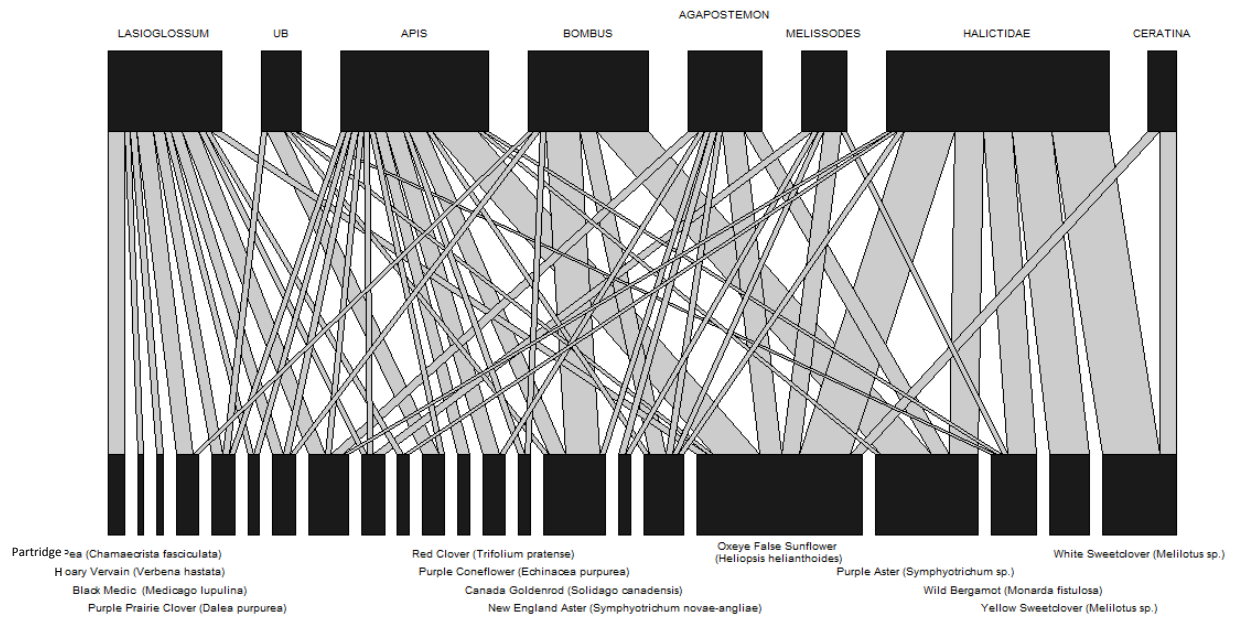
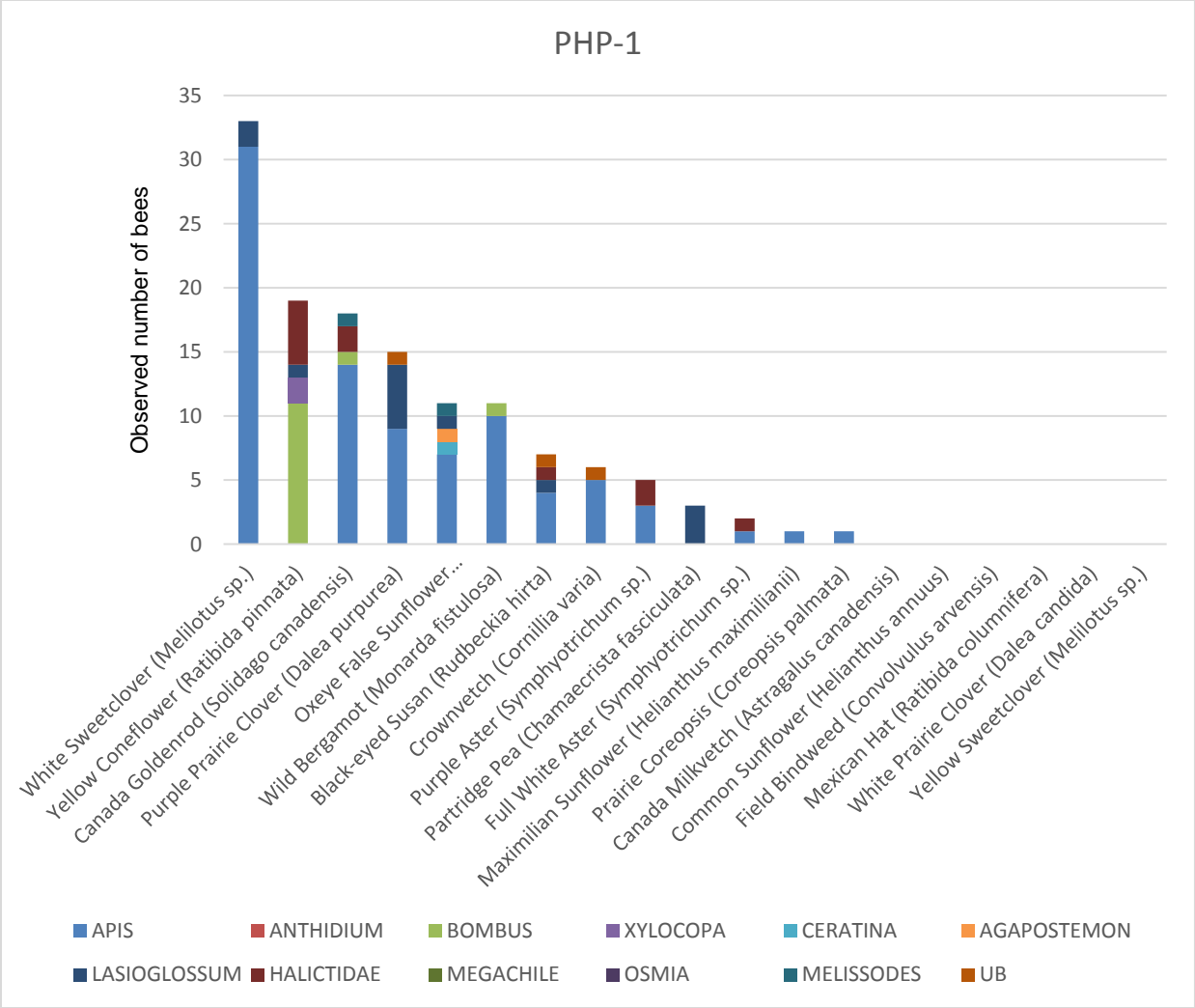


Figure 27. Bipartite interaction network between bee groups and plant species at site PHP-3.



**Figure 28. Abundance of groups of bees on plant species at site PHP-1. Plant species with no bar were present in the observation area (and in flower) but had no bees observed on them during the sampling periods.**

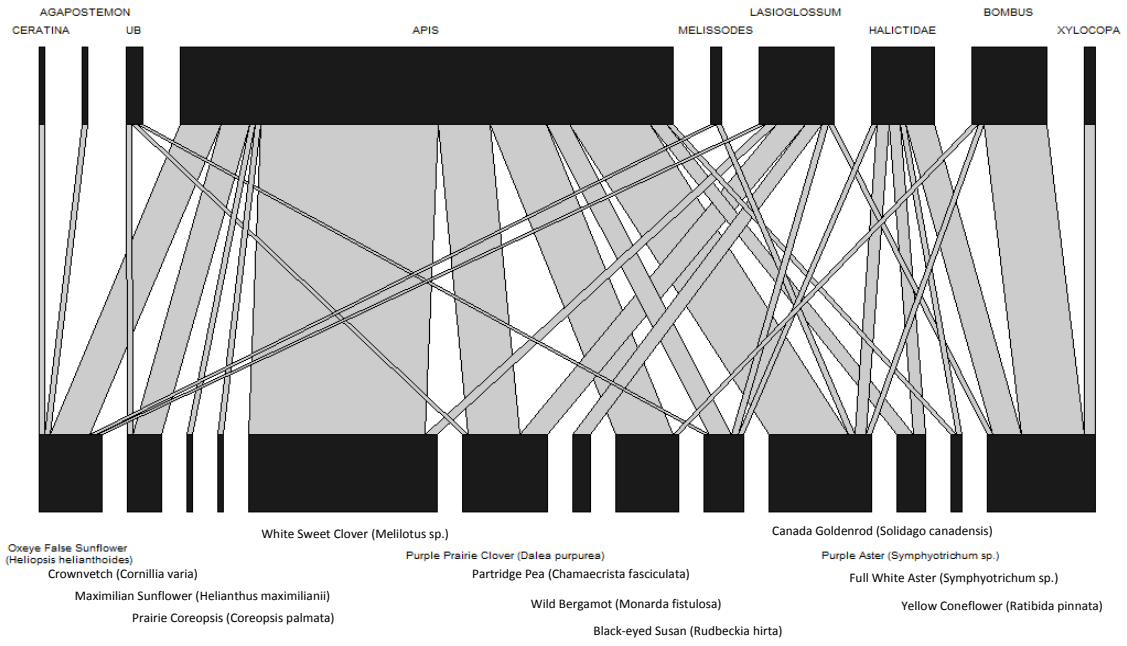


Figure 29. Bipartite interaction network between bee groups and plant species at site PHP-1.

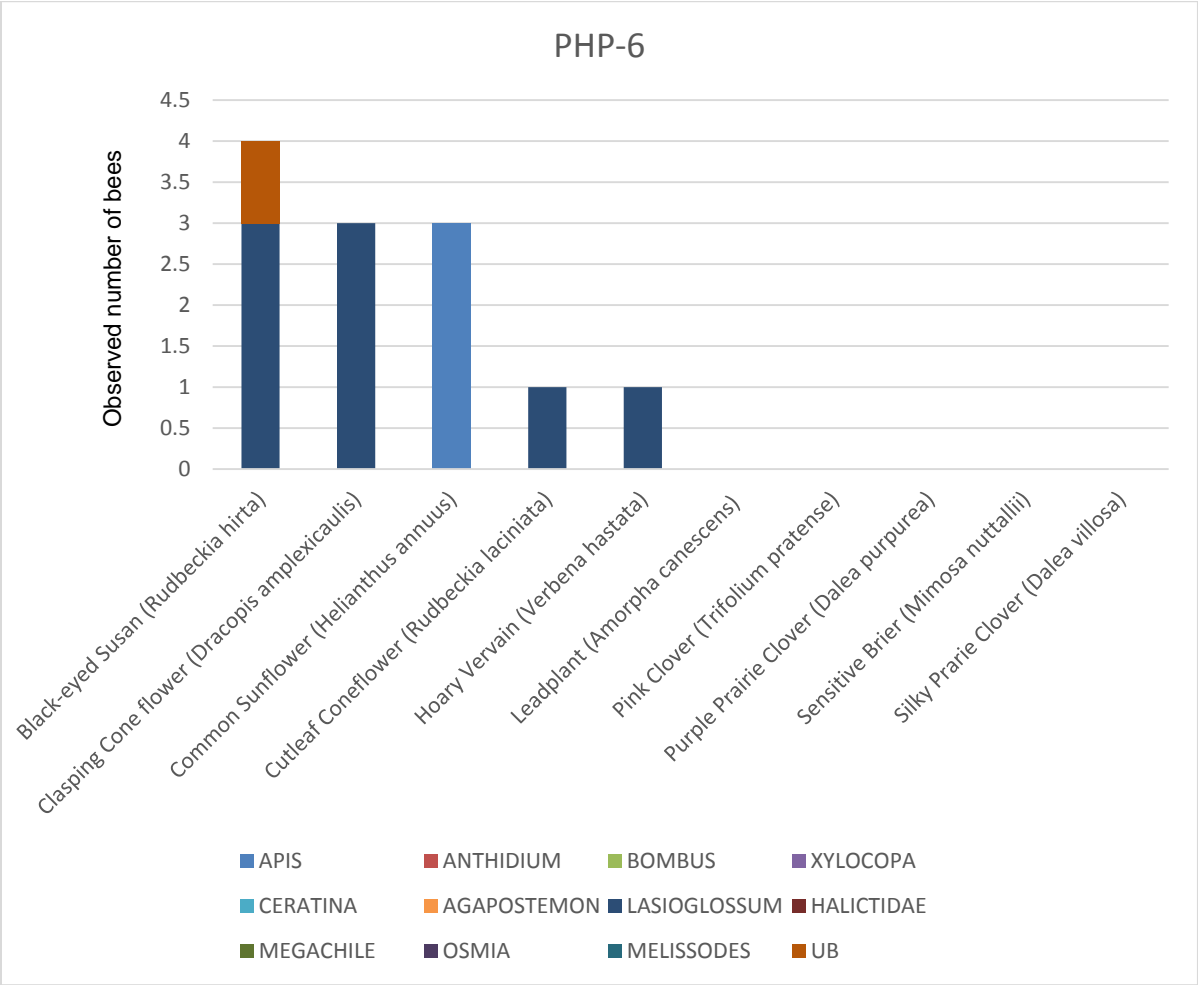


Figure 30. Abundance of groups of bees on plant species at site PHP-6. Plant species with no bar were present in the observation area (and in flower) but had no bees observed on them during the sampling periods.

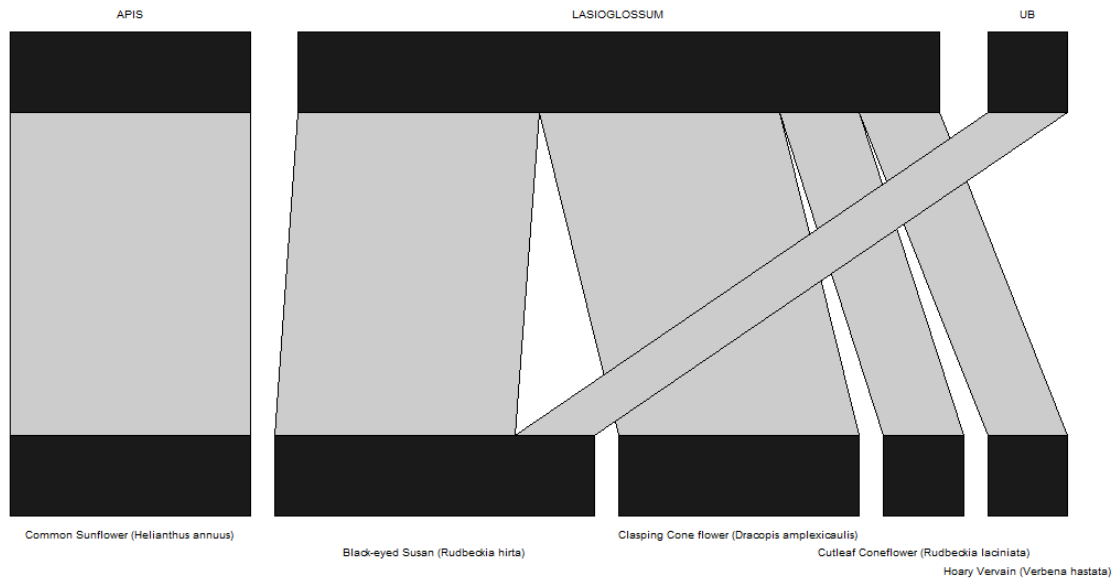
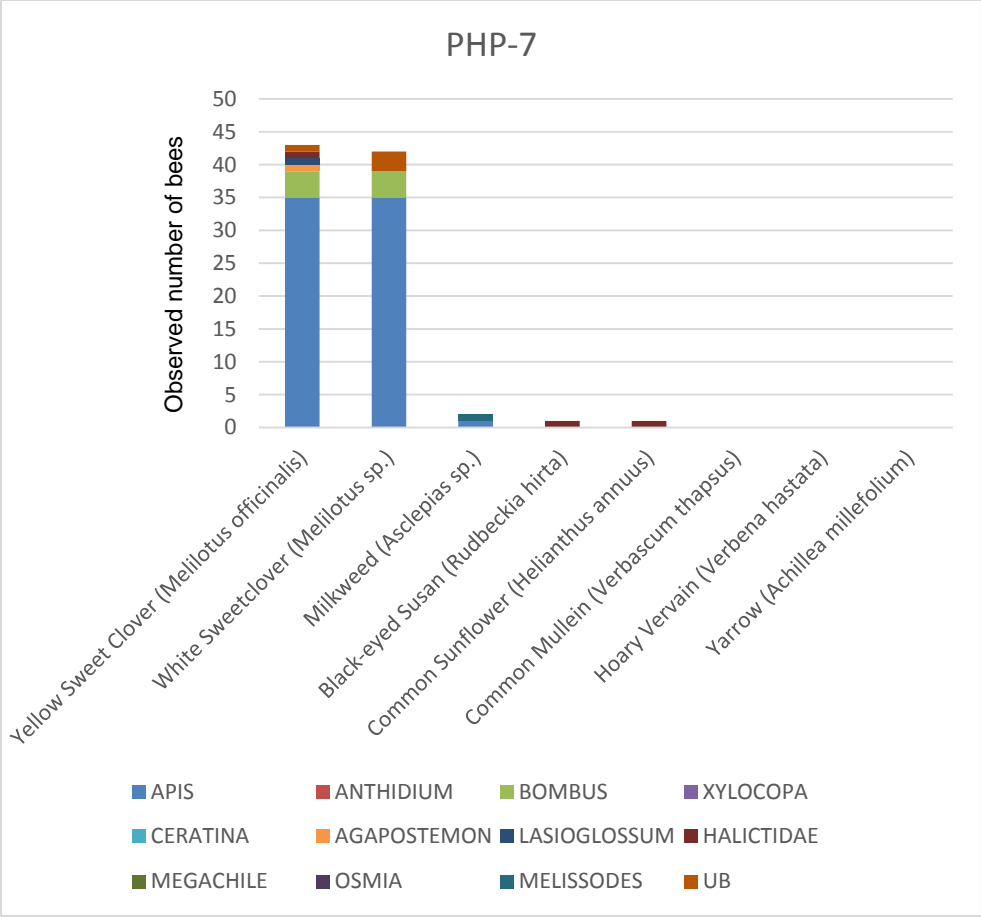


Figure 31. Bipartite interaction network between bee groups and plant species at site PHP-6.



**Figure 32. Abundance of groups of bees on plant species at site PHP-7. Plant species with no bar were present in the observation area (and in flower) but had no bees observed on them during the sampling periods.**



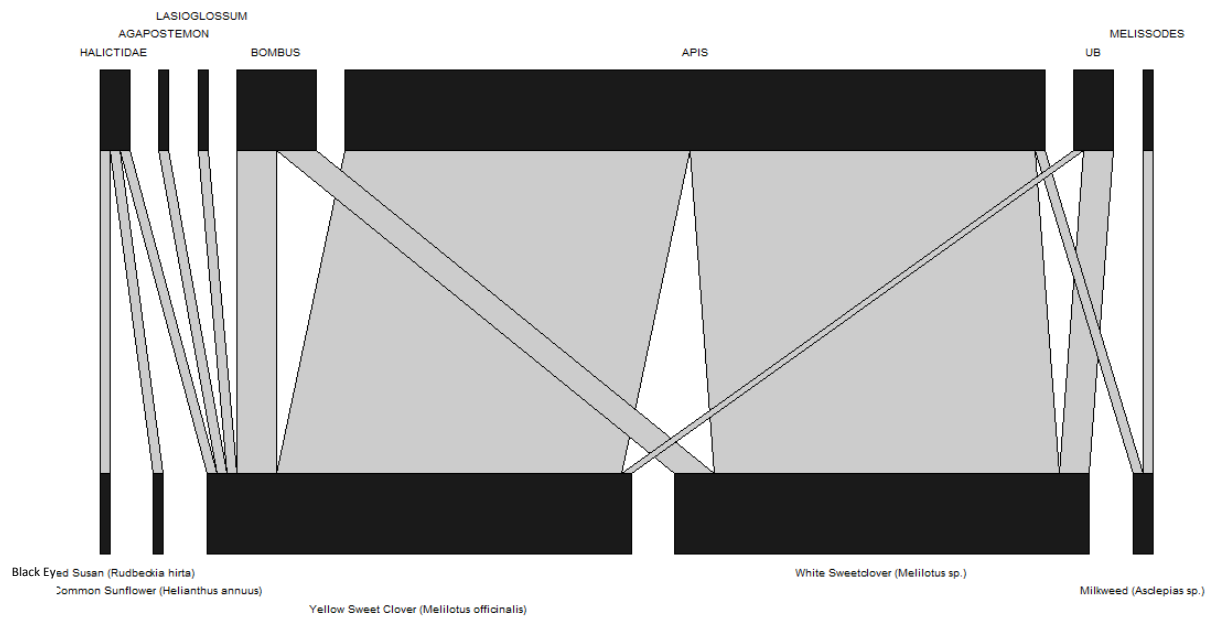
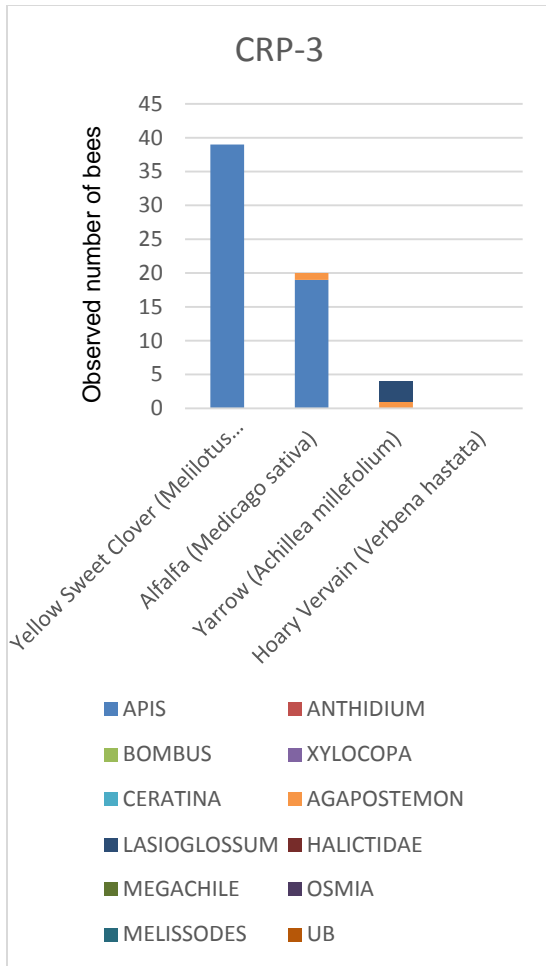


Figure 33. Bipartite interaction network between bee groups and plant species at site PHP-7.



**Figure 34. Abundance of groups of bees on plant species at site CRP-3. Plant species with no bar were present in the observation area (and in flower) but had no bees observed on them during the sampling periods.**

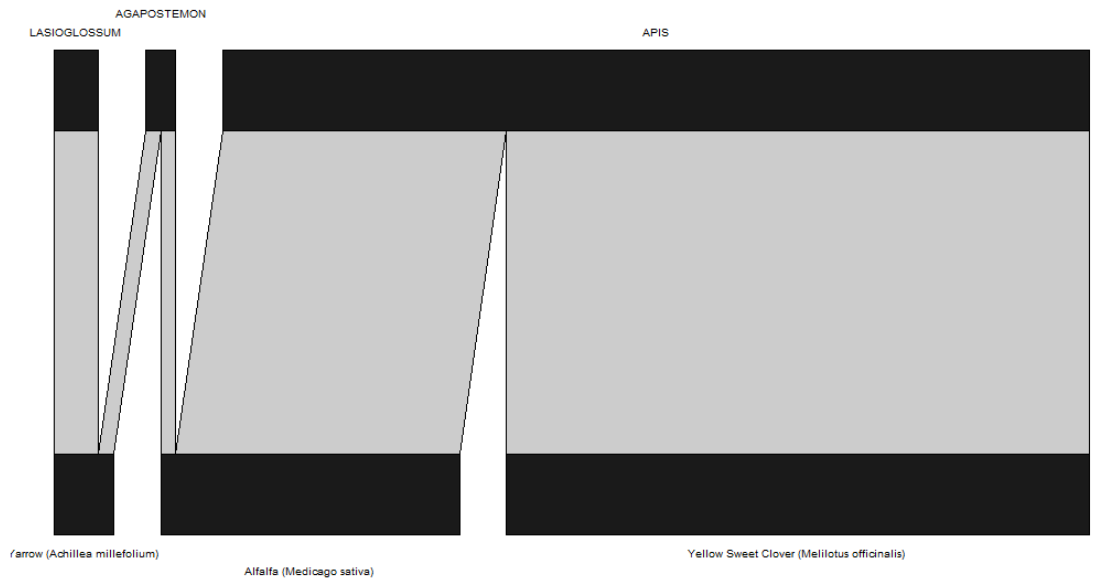


Figure 35. Bipartite interaction network between bee groups and plant species at site CRP-3.

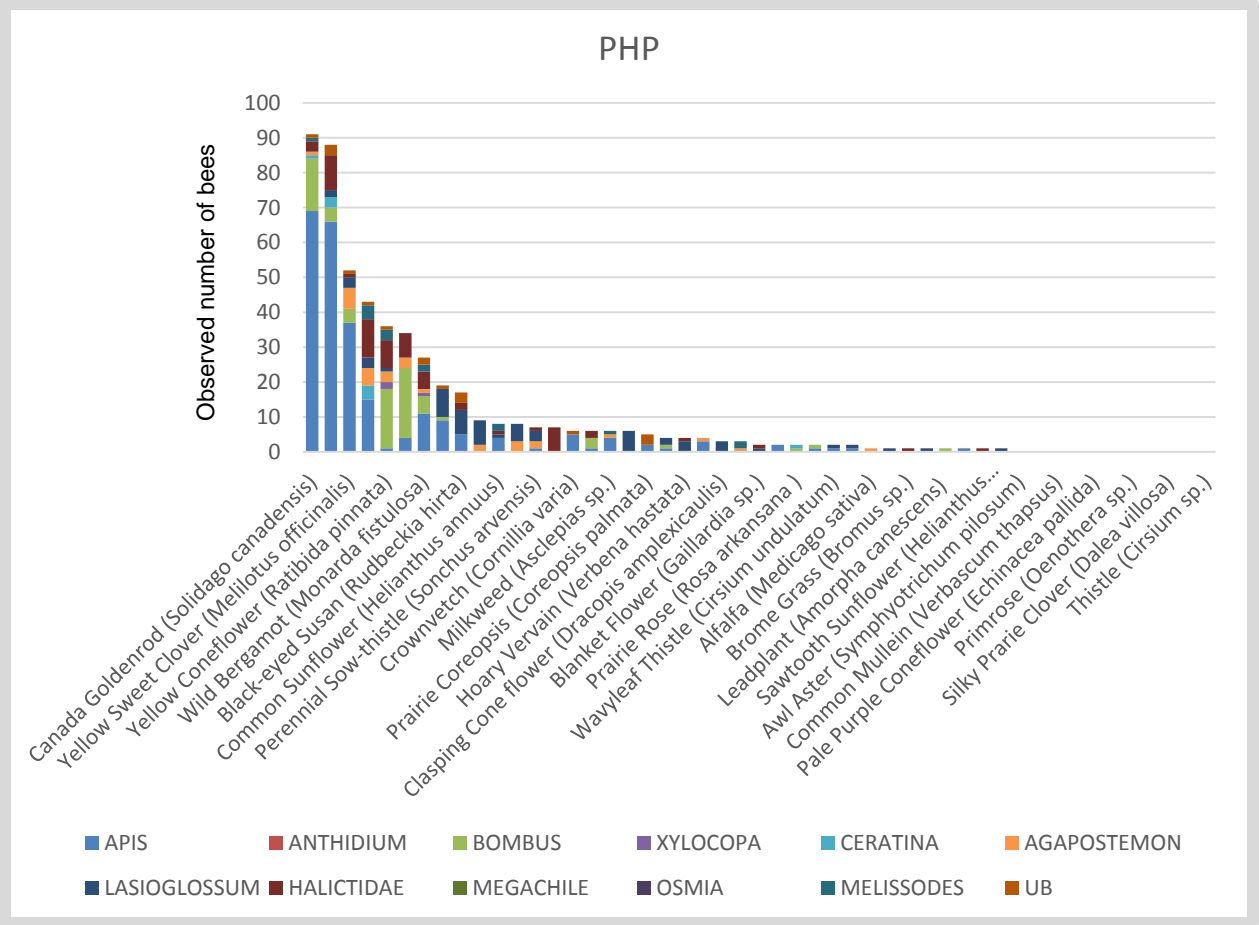


Figure 36. Abundance of bee groups on plant species pooled for six PHP sites. Plant species with no bar were present in the observation area (and in flower) but had no bees observed on them during the sampling periods.

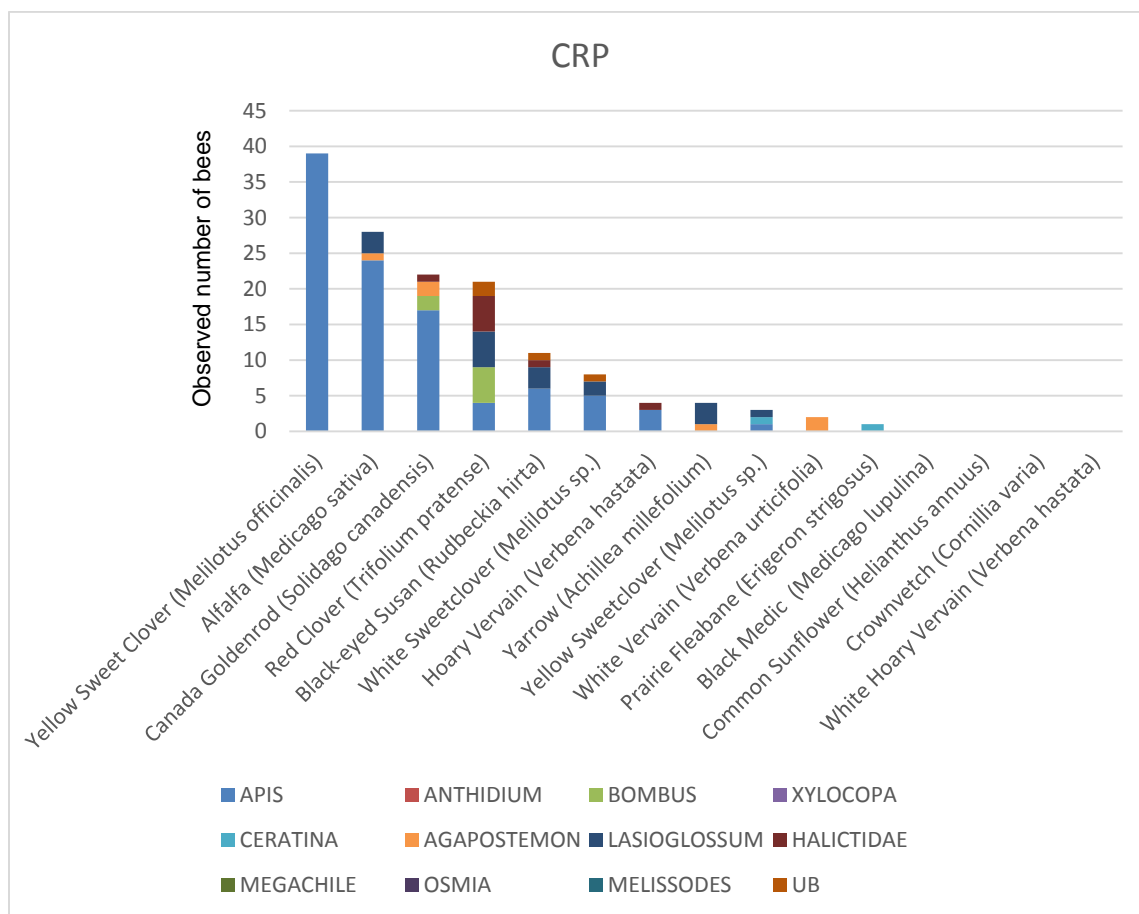


Figure 37. Abundance of bee groups on flower species pooled from two CRP sites. Plant species with no bar were present in the observation area (and in flower) but had no bees observed on them during the sampling periods.

Plant species in PHP were categorized on total bee abundance, bee group richness, total honey bee abundance, and total native bee abundance (Table 6). Plant species with the highest Plant Value Index based on these measure were (in order of descending score) Canada Goldenrod (*Solidago canadensis*), White Sweet Clover (*Melilotus albus*), Yellow Sweet Clover (*Melilotus officinalis*), Oxeye False Sunflower (*Heliopsis helianthoides*), Wild Bergamot (*Monarda fistulosa*), Purple Aster (*Symphyotrichum sp.*), Purple Prairie Clover (*Dalea purpurea*), Yellow Coneflower (*Ratibida pinnata*), Black-eyed Susan (*Rudbeckia hirta*), and Common Sunflower (*Helianthus annuus*). Based on this ranking system, these plants had the highest combined value for managed bees and native bee diversity and abundance support. These plants therefore provide the most value and it is recommended that they be included in seed mixes in areas where they are native, or similarly functioning native plants. White Sweet Clover and Yellow Sweet Clover were volunteers, not included in the PHP mixes, yet provided high value to pollinators. Decisions to include non-native,

high-value plants in PHP plantings can be addressed as the program goes forward. A number of plants included in the seed mixes received no bee visits during our observations. This does not mean that they are not a valuable resource to bees, having bee interactions outside of the relatively short time that they were observed. However, it is likely that they are not 'magnet' species with high value to many bees and/or a diversity of bees, although they could be important for less common species. Not including these plants in seed mixes may be a way to reduce seed costs (especially if they are relatively expensive seed), without significantly impacting the value of the plantings to bee communities.

**Table 6. Ranking of plants in CRP and PHP land based on abundance of bees observed and number of groups observed. Not shown is ranking by honey bee abundance and native bee abundance. All four rankings were summed to create a 'Bee Value Index' for each plant in PHP land.**

<b>Abundance</b>	<b>Group Richness</b>	<b>BEE VALUE RANK</b>	<b>VALUE</b>
Canada Goldenrod (Solidago canadensis)	Yellow Coneflower (Ratibida pinnata)	Canada Goldenrod (Solidago canadensis)	20
White Sweet Clover (Melilotus albus)	Canada Goldenrod (Solidago canadensis)	White Sweet Clover (Melilotus albus.)	20
Yellow Sweet Clover (Melilotus officinalis)	Oxeye False Sunflower (Heliopsis helianthoides)	Yellow Sweet Clover (Melilotus officinalis)	20
Oxeye False Sunflower (Heliopsis helianthoides)	Wild Bergamot (Monarda fistulosa)	Oxeye False Sunflower (Heliopsis helianthoides)	19
Yellow Coneflower (Ratibida pinnata)	White Sweet Clover (Melilotus albus)	Wild Bergamot (Monarda fistulosa)	18
Purple Aster (Symphyotrichum sp.)	Yellow Sweet Clover (Melilotus officinalis)	Purple Aster (Symphyotrichum sp.)	17
Wild Bergamot (Monarda fistulosa)	Black-eyed Susan (Rudbeckia hirta)	Purple Prairie Clover (Dalea purpurea)	17
Purple Prairie Clover (Dalea purpurea)	Common Sunflower (Helianthus annuus)	Yellow Coneflower (Ratibida pinnata)	17
Black-eyed Susan (Rudbeckia hirta)	Perennial Sow-thistle (Sonchus arvensis)	Black-eyed Susan (Rudbeckia hirta)	16
Salsify (Tragopogon sp.)	Purple Aster (Symphyotrichum sp.)	Common Sunflower (Helianthus annuus)	14
Common Sunflower (Helianthus annuus)	Purple Prairie Clover (Dalea purpurea)	Full White Aster (Symphyotrichum sp.)	13
Field Bindweed (Convolvulus arvensis)	Canada Milkvetch (Astragalus canadensis)	Milkweed (Asclepias sp.)	13
Perennial Sow-thistle (Sonchus arvensis)	Full White Aster (Symphyotrichum sp.)	Perennial Sow-thistle (Sonchus arvensis)	13
Crownvetch (Cornillia varia)	Milkweed (Asclepias sp.)	Canada Milkvetch (Astragalus canadensis)	12
Full White Aster (Symphyotrichum sp.)	Blanket Flower (Gaillardia sp.)	Crownvetch (Cornillia varia)	12

Milkweed ( <i>Asclepias</i> sp.)	Crownvetch ( <i>Cornillia varia</i> )	Field Bindweed ( <i>Convolvulus arvensis</i> )	12
Partridge Pea ( <i>Chamaecrista fasciculata</i> )	Field Bindweed ( <i>Convolvulus arvensis</i> )	Prairie Coreopsis ( <i>Coreopsis palmata</i> )	12
Prairie Coreopsis ( <i>Coreopsis palmata</i> )	Hoary Vervain ( <i>Verbena hastata</i> )	Salsify ( <i>Tragopogon</i> sp.)	12
Canada Milkvetch ( <i>Astragalus canadensis</i> )	New England Aster ( <i>Symphyotrichum novae-angliae</i> )	Red Clover ( <i>Trifolium pratense</i> )	11
Hoary Vervain ( <i>Verbena hastata</i> )	Prairie Coreopsis ( <i>Coreopsis palmata</i> )	Hoary Vervain ( <i>Verbena hastata</i> )	10
Red Clover ( <i>Trifolium pratense</i> )	Prairie Rose ( <i>Rosa arkansana</i> )	New England Aster ( <i>Symphyotrichum novae-angliae</i> )	10
Clasping Cone flower ( <i>Dracopis amplexicaulis</i> )	Purple Coneflower ( <i>Echinacea purpurea</i> )	Partridge Pea ( <i>Chamaecrista fasciculata</i> )	10
New England Aster ( <i>Symphyotrichum novae-angliae</i> )	Red Clover ( <i>Trifolium pratense</i> )	Purple Coneflower ( <i>Echinacea purpurea</i> )	10
Blanket Flower ( <i>Gaillardia</i> sp.)	Salsify ( <i>Tragopogon</i> sp.)	Wavyleaf Thistle ( <i>Cirsium undulatum</i> )	10
Canada Thistle ( <i>Cirsium arvense</i> )	Wavyleaf Thistle ( <i>Cirsium undulatum</i> )	White Prairie Clover ( <i>Dalea candida</i> )	10
Prairie Rose ( <i>Rosa arkansana</i> )	White Prairie Clover ( <i>Dalea candida</i> )	Blanket Flower ( <i>Gaillardia</i> sp.)	9
Purple Coneflower ( <i>Echinacea purpurea</i> )	Alfalfa ( <i>Medicago sativa</i> )	Clasping Cone flower ( <i>Dracopis amplexicaulis</i> )	9
Wavyleaf Thistle ( <i>Cirsium undulatum</i> )	Black Medic ( <i>Medicago lupulina</i> )	Prairie Rose ( <i>Rosa arkansana</i> )	9
White Prairie Clover ( <i>Dalea candida</i> )	Brome Grass ( <i>Bromus</i> sp.)	Canada Thistle ( <i>Cirsium arvense</i> )	8
Alfalfa ( <i>Medicago sativa</i> )	Canada Thistle ( <i>Cirsium arvense</i> )	Alfalfa ( <i>Medicago sativa</i> )	7
Black Medic ( <i>Medicago lupulina</i> )	Clasping Cone flower ( <i>Dracopis amplexicaulis</i> )	Black Medic ( <i>Medicago lupulina</i> )	7
Brome Grass ( <i>Bromus</i> sp.)	Cutleaf Coneflower ( <i>Rudbeckia laciniata</i> )	Brome Grass ( <i>Bromus</i> sp.)	7
Cutleaf Coneflower ( <i>Rudbeckia</i> )	Leadplant ( <i>Amorpha canescens</i> )	Cutleaf Coneflower ( <i>Rudbeckia</i> )	7



laciniata)		laciniata)	
	Maximilian Sunflower		
Leadplant ( <i>Amorpha canescens</i> )	( <i>Helianthus maximiliani</i> )	Leadplant ( <i>Amorpha canescens</i> )	7
Maximilian Sunflower	Partridge Pea ( <i>Chamaecrista</i>	Maximilian Sunflower	
( <i>Helianthus maximiliani</i> )	<i>fasciculata</i> )	( <i>Helianthus maximiliani</i> )	7
Sawtooth Sunflower ( <i>Helianthus</i>	Sawtooth Sunflower ( <i>Helianthus</i>	Sawtooth Sunflower	
<i>grosseserratus</i> )	<i>grosseserratus</i> )	( <i>Helianthus grosseserratus</i> )	7
Yarrow ( <i>Achillea millefolium</i> )	Yarrow ( <i>Achillea millefolium</i> )	Yarrow ( <i>Achillea millefolium</i> )	7
Awl Aster ( <i>Symphotrichum</i>	Awl Aster ( <i>Symphotrichum</i>	Awl Aster ( <i>Symphotrichum</i>	
<i>pilosum</i> )	<i>pilosum</i> )	<i>pilosum</i> )	4
Common Milkweed ( <i>Asclepias</i>	Common Milkweed ( <i>Asclepias</i>	Common Milkweed ( <i>Asclepias</i>	
<i>syriaca</i> )	<i>syriaca</i> )	<i>syriaca</i> )	4
Common Mullein ( <i>Verbascum</i>	Common Mullein ( <i>Verbascum</i>	Common Mullein ( <i>Verbascum</i>	
<i>thapsus</i> )	<i>thapsus</i> )	<i>thapsus</i> )	4
Mexican Hat ( <i>Ratibida</i>	Mexican Hat ( <i>Ratibida</i>	Mexican Hat ( <i>Ratibida</i>	
<i>columnifera</i> )	<i>columnifera</i> )	<i>columnifera</i> )	4
Pale Purple Coneflower	Pale Purple Coneflower	Pale Purple Coneflower	
( <i>Echinacea pallida</i> )	( <i>Echinacea pallida</i> )	( <i>Echinacea pallida</i> )	4
		Pink Clover ( <i>Trifolium</i>	
Pink Clover ( <i>Trifolium pratense</i> )	Pink Clover ( <i>Trifolium pratense</i> )	<i>pratense</i> )	4
Primrose ( <i>Oenothera</i> sp.)	Primrose ( <i>Oenothera</i> sp.)	Primrose ( <i>Oenothera</i> sp.)	4
		Sensitive Brier ( <i>Mimosa</i>	
Sensitive Brier ( <i>Mimosa nuttallii</i> )	Sensitive Brier ( <i>Mimosa nuttallii</i> )	<i>nuttallii</i> )	4
		Silky Prarie Clover ( <i>Dalea</i>	
Silky Prarie Clover ( <i>Dalea villosa</i> )	Silky Prarie Clover ( <i>Dalea villosa</i> )	<i>villosa</i> )	4
Spiderwort ( <i>Tradescantia</i> sp.)	Spiderwort ( <i>Tradescantia</i> sp.)	Spiderwort ( <i>Tradescantia</i> sp.)	4
Thistle ( <i>Cirsium</i> sp.)	Thistle ( <i>Cirsium</i> sp.)	Thistle ( <i>Cirsium</i> sp.)	4

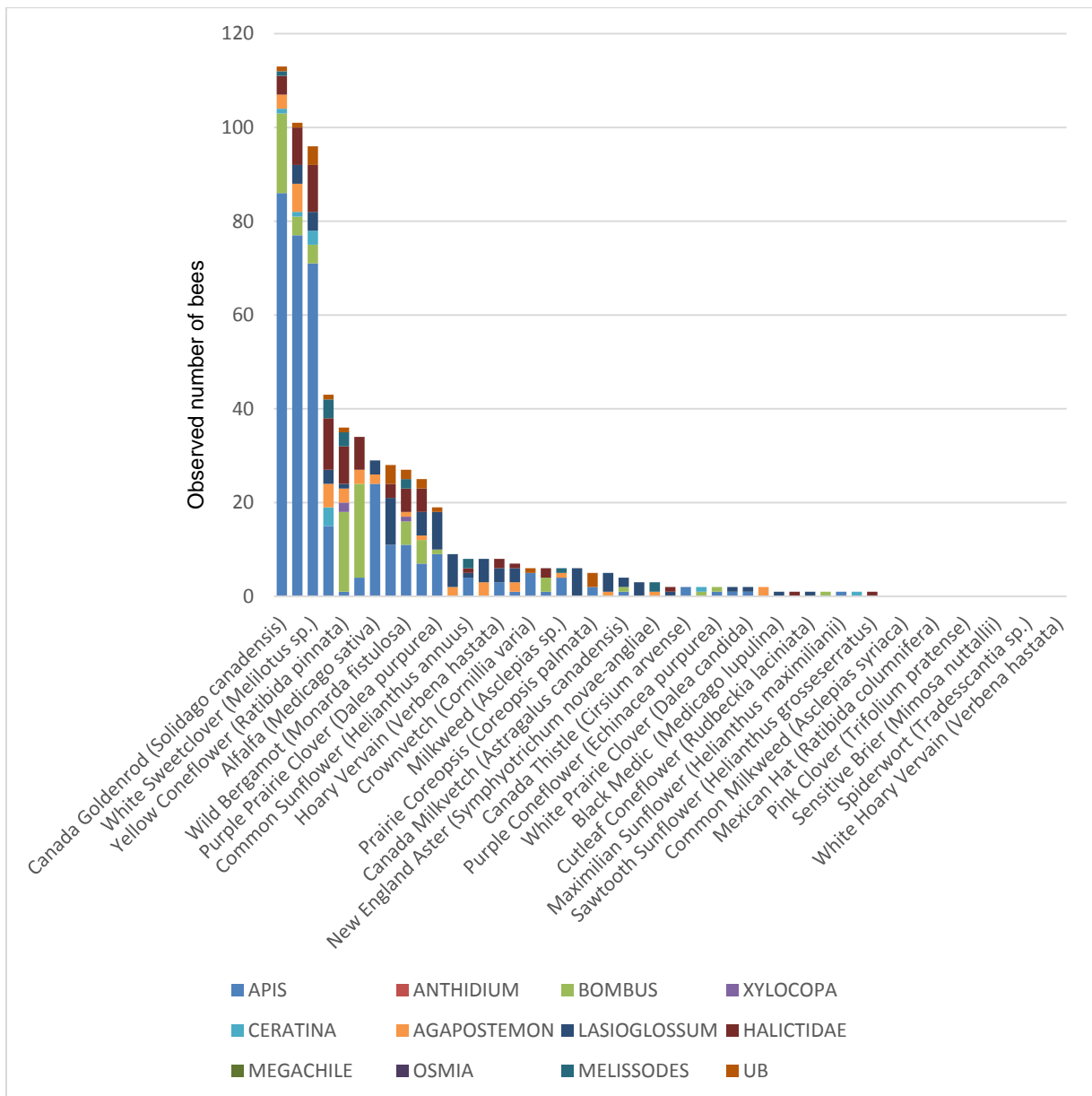


Figure 38. Pooled CRP and PHP plant-bee interactions presenting in descending Plant Value Index order. Plant species with no bar were present in the observation area (and in flower) but had no bees observed on them during the sampling periods.

### Native bee community characteristics

A total of 29 unique bee genera were identified across all states and landscapes sampled. PHP landscapes had 28 of these genera, while CRP and crop landscapes had 15. Nine of these bee genera were present only on PHP landscapes (Table 7). One genera was only seen on CRP lands. Of the genera present on PHP lands four (*Tripeolus*, *Oreopasites*, *Epeolus*, and *Coelioxys*) are cleptoparasitic, meaning they do not collect pollen, but instead lay either eggs in the nest of other bees, notably in the genus *Megachile* (Table 7). The presence of parasites in ecosystems documents trophic complexity.

**Table 7. List of bee families and genera identified in samples from all states and across all sites. \* genus unique to PHP, \*\* genus unique to CRP(1 or 2). Size: S= 6 mm or less, M= 7-10mm ,L=11 mm or larger. Feeding: P=polylectic, M=meselectic, O=oligolectic, CP= cleptoparasite.**

Family	Genus	Bee life history characteristics			Landscape		
		Size	Feeding	Nesting	Crop	CRP	CP-42
Andrenidae	Andrena	L	M	ground		X	X
Andrenidae	Calliopsis	L	M	ground	X		X
Andrenidae	Protandrena**	L	M	ground		X	
Apidae	Anthophora	L	O	ground		X	X
Apidae	Apis	L	P	Colonial, hive	X	X	X
Apidae	Bombus	L	P	Colonial, ground/cavity		X	X
Apidae	Ceratina	S	O	ground	X		X
Apidae	Diadasia	L	M	ground	X		X
Apidae	Epeolus*	M	O	ground			X
Apidae	Exomalopsis*	M	O	ground			X
Apidae	Melissodes	M	O	ground	X	X	X
Apidae	Oreopasites*	M	CP	Cleptoparasite			X
Apidae	Svastra	L	M	ground	X	X	X
Apidae	Triepeolus*	L	CP	Cleptoparasite			X
Colletidae	Colletes	M	M	ground	X		X
Halictidae	Agapostemon	M	O	ground	X	X	X
Halictidae	Auglochlorella	M	O	ground	X	X	X
Halictidae	Dialictus	S	O	ground	X	X	X
Halictidae	Halictus	L	O	ground	X	X	X
Halictidae	Lasioglossum	S	O	ground	X	X	X
Halictidae	Sphecodes	S	O	ground	X		X
Megachilidae	Anthidium*	L	O	cavity			X
Megachilidae	Ashmeadiella*	M	O	cavity			X
Megachilidae	Coelioxys*	L	CP	Cleptoparasite			X
Megachilidae	Hoplitis*	M	CP	Cleptoparasite			X
Megachilidae	Megachile	L	O	cavity	X	X	X
Megachilidae	Osmia	M	O	cavity		X	X
Megachilidae	Dianthidium	L	O	cavity	X	X	X
Melittidae*	Hesperapis*	L	O	ground			X

### Cost and Partial Benefit Analysis

PHP is more costly per acre than CRP. Mean PHP enhancements within CRP land was \$264 greater (including SIP), more per acre than CRP alone (mean \$295 vs. \$31) (Table 9). PHP program costs are higher due to greater seed mix costs and SIP associated with the practice, and we include this total greater cost in the following analysis because all sites in our study include SIP. We did not parse out who bears the cost or incurs the benefit. Without the SIP payment, the additional cost of the program over CRP alone averages \$145.

Whether the greater cost of the program can be offset by direct economic benefits (e.g. greater honey production) and/or ecosystem services is largely unknown. However, partial economic analysis, using enhanced honey production demonstrates that on PHP land there was, on average, 33 lbs more honey per colony (72 lbs CRP vs 105 on PHP), a 42% greater production per colony (Table 8). Using the program cost (seed plus SIP) for the area of each site, this translates to a honey production cost of \$30/lb at CRP sites and \$10/lb at PHP sites, a difference of \$20/lb (Table 8). The greater cost of PHP is offset by the smaller size and greater honey production potential. However, it is noted that honey bees use land on a large scale, and honey bees likely were using a much wider areas of the landscape than what was seeded in PHP. But, because surrounding land had a similar composition for both site types, differences in honey production between solely CRP lands and CRP with PHP enhancements, likely were directly due to the PHP enhancement.

Commercial price for extracted, unprocessed honey in major producing states by packers, handlers & other large users varies, but generally is in the range of \$2/lb (<https://search.ams.usda.gov/mnreports/fvmhoney.pdf>). The increase in honey therefore, for each colony in PHP lands, translates to approximately \$66 extra per colony.

Whether this increased honey production can be supported over a large number of colonies, and at what number of colonies PHP land value would no longer provide an increased benefit to honey production, due to exploitation of all added resources, is not known. The value cannot be translated to a per acre benefit of PHP to honey production because number of colonies and acreage varied among sites, and it is unknown how much of the PHP enhancement was needed for the greater honey production. However, putting the numbers into a per acre can be somewhat useful for assessment of benefits. For example, if one acre of PHP could support the enhanced honey production seen in this study (33lbs extra honey) for 3-4 honey bee colonies, the enhanced honey production would completely offset the cost of the \$245/acre extra cost for PHP. However, all of the PHP area may have been needed for the increased honey production observed. It is likely it would take multiple acres of to support 3-4 honey bee colonies and greater honey production from the enhanced resources, and therefore costs would also increase. Yet, these values show that, to some extent, increased costs can be directly offset by increased honey production. Note that this is a hypothetical situation and only meant to show possible cost offsets and would vary by amount of added resources in PHP land, number of honey bee colonies, surrounding land matrix, strength of colonies, and price of honey amongst other factors.

It is not possible at this time to calculate monetary value of enhanced native bees and other wildlife on PHP land (although economic quantification is possible in some situations such as agricultural

crops that benefits from native bee pollination). There can however be a ‘price’ calculated for the enhanced bee communities seen on PHP land. Whether the extra ‘price’ of more bees or greater bee diversity is justified and valued will depend on the individual or group valuing the enhancement, or, future direct economic benefit analyses. It was found that in PHP sites there were, on average, 23 additional individual bees relative to the same sampling protocol on CRP sites (37% increase in bee abundance), and 5.1 more bee taxa (70% increase) compared to the average for CRP sites (Table 9). The ‘price’ therefore for the enhanced bee taxa of 63% and enhanced bee abundance of 31% is \$245/acre. Valuation in economic terms such as ecosystem services is not possible at this time, however, the enhanced bee diversity and abundance will contribute to ecosystem function and services on the land and surrounding land within bee foraging ranges. It is important to note that many other potential benefits such as wildlife enhancement, and ecosystem services to agriculture also offset cost if they are valued.

**Table 8: Costs and benefits seen for honey bee productivity measured in mean annual pounds of honey produced between CRP and PHP sites (with SIP). PHP-CRP shows the difference in costs and honey production. Note that acreage generally is greater in CRP lands (and therefore costs are greater); however, the results are presented in absolute values because honey production overall, regardless of size of habitat is the measured partial benefit of PHP enrollment. Note that this is a partial valuation excluding many potential benefits of PHP enhancements, discussed further in the text.**

<b>CRP</b>	<b>Cost of total enrollment</b>	<b>Pounds of Honey - mean annual</b>	<b>Cost of one pound of honey</b>	<b>Pounds of honey per dollar</b>
Iowa	\$1,115.39	95.3	\$11.70	0.09
Nebraska	\$201.12	50.1	\$4.02	0.25
Montana	\$5,112.93	70.5	\$72.54	0.01
<b>CRP Mean</b>	<b>\$2,143.15</b>	<b>71.9</b>	<b>\$29.79</b>	<b>0.03</b>
<b>PHP</b>	<b>Cost of total enrollment</b>	<b>Pounds of Honey - mean annual</b>	<b>Cost of one pound of honey</b>	<b>Pounds of honey per dollar</b>
Iowa	\$1,779.19	136.8	\$13.00	0.08
Nebraska	\$870.82	69.0	\$12.63	0.08
Montana	\$420.95	109.6	\$3.84	0.26
<b>PHP Mean</b>	<b>\$1,023.65</b>	<b>105.1</b>	<b>\$9.74</b>	<b>0.10</b>
<b>PHP - CRP</b>	<b>-\$1,119.50</b>	<b>33.17</b>	<b>-\$20.05</b>	<b>0.07</b>

**Table 9. Mean costs for PHP and CRP, and bee richness (family, genus, taxon level) and abundance measure (bees per sample period in each state). The percentage increase in bee community variables associated with PHP is compared to the mean cost differential of the two practices at the sites in this study, to get a 'price' in dollars of a 1% increase in that variable. Note that this is a partial analysis of benefits excluding many potential benefits of PHP enhancements, discussed further in the text.**

<b>CRP</b>	<b>Acres enrolled</b>	<b>Cost of enrollment per acre</b>	<b>No. of families</b>	<b>No. of Genera</b>	<b>No. lowest taxa</b>	<b>Abundance</b>
IA	13.6	\$82.01	2	4	4	33
NE	15	\$13.41	3	8	11	86
MT	202	\$25.31	3	7.5	9	37
WA	262.6	\$3.08	1.5	4	5.5	89
<b>Mean CRP</b>	<b>123.3</b>	<b>\$30.95</b>	<b>2.3</b>	<b>5.8</b>	<b>7.3</b>	<b>61.7</b>
<b>PHP</b>	<b>Acres enrolled</b>	<b>Cost of enrollment per acre</b>	<b>No. of families</b>	<b>No. of Genera</b>	<b>No. lowest taxa</b>	<b>Relative abundance</b>
IA	5.27	\$337.61	2.3	5.3	5.33	29
NE	4.03	\$216.08	3	11	14	67
MT	1.35	\$311.81	3	4.5	5.5	16
WA	3.4	\$238.79	3.3	12	23.3	208
<b>Mean PHP</b>	<b>4.3</b>	<b>\$295.05</b>	<b>2.9</b>	<b>8.4</b>	<b>12.4</b>	<b>84.5</b>
<b>PHP-CRP</b>		<b>\$264.10</b>	<b>0.60</b>	<b>2.60</b>	<b>5.10</b>	<b>22.80</b>
<b>Percent increase in PHP</b>			<b>26.1</b>	<b>44.8</b>	<b>69.9</b>	<b>37.0</b>
<b>Dollars/% increase</b>			<b>\$9.39</b>	<b>\$5.47</b>	<b>\$3.51</b>	<b>\$6.63</b>

### **Conclusions and Recommendations**

Of the variables tested in this study, the categorical variable of enrollment in PHP had a significant correlation with increased honey bee hive productivity and increased native bee abundance. Increasing PHP enrollment size was also significant in explaining increasing patterns of native bee occurrence. The variation in PHP seeding size available for this study ranged from 0.9 acres to 10.5 acres, significantly smaller than CRP reenrollment sizes. This comparatively small component of the CRP landscape has a significant impact in supporting native bee communities and honey bee productivity. Although more expensive, the conservation and economic benefits of PHP were demonstrated by greater honey production and more diverse and abundant bee communities, and we recommend continuing to expand PHP enrollment acres. Outreach programs, encouraging PHP enrollments, should be created and employed in areas where there has been low uptake of PHP.

Assessments of plant-pollinator interactions indicate that specific floral components in the seed mix are responsible for most of the pollinator benefits observed in this study and can be used to assess seed mix components and optimization of costs and benefits. The area-related response of the native bee community to PHP seeding is likely correlated to the increased presence of native plants and increased food resources, although not explicitly quantified in this study. PHP area-related support trends result from data aggregated from the four states and can be used broadly to describe impacts for production systems within the Midwest (corn) and the plains (wheat). We present a floral species list that describes the biological component of increased pollinator support. Because eco-regionally specific native plants are used in each PHP mix, our bee-plant interaction results speak specifically to landscapes within the Midwest, and more particularly to Iowa and Nebraska. Similar trends are expected in all states and landscapes with PHP enrollment, but the plant species driving these trends cannot be predicted. We can suggest that native plant species within the same genus should act in a similar manner, however further recommend assessing bee-plant networks across all CRP regions.



## References

**Aizen, M.A., Garibaldi, L.A., Cunningham, S.A., and Klein, A.M., 2009, How much does agriculture depend on pollinators? Lessons from long-term trends in crop production: *Annals of Botany*, v. 103, no. 9, p. 1579–1588.**

Productivity of many crops benefits from the presence of pollinating insects, so a decline in pollinator abundance should compromise global agricultural production. Motivated by the lack of accurate estimates of the size of this threat, we quantified the effect of total loss of pollinators on global agricultural production and crop production diversity. The change in pollinator dependency over 46 years was also evaluated, considering the developed and developing world separately.

**Bauer, D.M., and Wing, I.S., 2010, Economic consequences of pollinator declines—A synthesis: *Agricultural and Resource Economics Review*, v. 39, no. 3, p. 368–383.**

This paper surveys the literature on pollinator declines and related concerns regarding global food security. Methods for valuing the economic risks associated with pollinator declines are also reviewed. A computable general equilibrium (CGE) approach is introduced to assess the effects of a global catastrophic loss of pollinators. There appears to be evidence supporting a trend towards future pollinator shortages in the United States and other regions of the world. Results from the CGE model show economic risks to both direct crop sectors and indirect noncrop sectors in the economy, with some amount of regional heterogeneity.

**Best, L.B., Campa III, H., Kemp, K.E., Robel, R.J., Ryan, M.R., Savidge, J.A., Weeks Jr., H.P., and Winterstein, S.R., 1997, Bird abundance and nesting in CRP fields and cropland in the Midwest—A regional approach: *Wildlife Society Bulletin*, v. 25, no. 4, p. 864–877.**

We compared the abundance and nesting success of avian species in Conservation Reserve Program (CRP) fields during the summer with that in rowcrop fields over 5 years (1991-1995) for 6 midwestern states (Ind., Ia., Kans., Mich., Mo., and Nebr.). Field techniques were standardized in all states. CRP fields consisted of either perennial introduced grasses and legumes (CP1) or perennial native grasses (CP2), and the plant species seeded in CRP fields differed within and among the states. Disturbances to CRP fields included mowing (partial or complete), application of herbicides, and burning. The height, vertical density, and canopy coverage of vegetation in CRP fields were measured in each state; values for these measurements were particularly low in Kansas. Mean annual total bird abundance in CRP fields ranged from 4.9 to 29.3 birds/km of transect. The most abundant species on CRP fields differed among states but included red-winged blackbirds (*Agelaius phoeniceus*), grasshopper sparrows (*Ammodramus savannarum*), and dickcissels (*Spiza americana*). Although the total number of bird species was similar in CRP and rowcrop fields across the region, bird abundance was 1.4-10.5 times greater in the former. Nests of 33 bird species were found in CRP fields compared with only 10 species in rowcrop fields, and the number of nests found was 13.5 times greater in CRP fields. Nest success in CRP fields was 40% overall; predation was the greatest cause of nest failure. Long-term farm set-aside programs that establish perennial grass cover, such as the CRP, seem to provide many benefits for grassland birds, including several species for which conservation is a great concern.

**Blaauw, B. R. and Isaacs, R. (2014), Flower plantings increase wild bee abundance and the pollination services provided to a pollination-dependent crop. *J Appl Ecol*, 51: 890–898. doi:10.1111/1365-2664.12257**

1. Pollination services from wild insects contribute to crop productivity around the world, but are at risk of decline in agricultural landscapes. Using highbush blueberry as a model system, we tested whether wildflower plantings established adjacent to crop fields would increase the abundance of wild pollinators during crop bloom and enhance pollination and yield.
2. Plantings were seeded in 2009 with a mix of 15 perennial wildflower species that provided season-long bloom and increased plant density and floral area during the subsequent 3 years.
3. Honeybees visiting blueberry flowers had similar abundance in enhanced and control fields in all 4 years of this study, whereas wild bee and syrphid abundance increased annually in the fields adjacent to wildflower plantings.

4. Crop pollination parameters including percentage fruit set, berry weight and mature seeds per berry were significantly greater in fields adjacent to wildflower plantings 3 and 4 years after seeding, leading to higher crop yields and with the associated revenue exceeding the cost of wildflower establishment and maintenance.
5. *Synthesis and applications.* We suggest that provision of forage habitat for bees adjacent to pollinator-dependent crops can conserve wild pollinators in otherwise resource-poor agricultural landscapes. Over time, these plantings can support higher crop yields and bring a return on the initial investment in wildflower seed and planting establishment, also insuring against loss of managed pollinators. Further understanding of the importance of planting size, location and landscape context will be required to effectively implement this practice to support crop pollination.

**Bollinger, E.K., and Gavin, T.A., 1989, Eastern bobolink populations—Ecology and conservation in an agricultural landscape, in Hagen, J.M., III and Johnston, D.W., eds., Ecology and conservation of neotropical migrant landbirds: Washington, D.C., Smithsonian Institute Press, p. 497–506.**

**Burger, L.W. Jr., Kurzejeski, E.W., Dailey, T.V., and Ryan, M.R., 1993, Relative invertebrate abundance and biomass in Conservation Reserve Program plantings in northern Missouri, in Church, K.E., and Dailey, T.V., eds., Quail III—National Quail Symposium: Jefferson City, Mo., Missouri Department of Conservation.**

We measured relative invertebrate abundance, biomass, and diversity in Conservation Reserve Program (CRP) fields planted to red clover (*Trifolium pratense*)/timothy (*Phleum pratense*), timothy, orchard-grass (*Dactylis glomerata*), tall fescue (*Festuca pratensis*), warm-season grasses (big bluestem [*Andropogon gerardi*]/switch grass [*Panicum virgatum*]), orchard-grass/Korean lespedeza (*Kummerowia stipu/acea*), and conventionally-tilled soybeans, to assess brood habitat quality for northern bobwhite (*Colinus virginianus*). We sampled invertebrate populations by vacuuming along 3 15-m transects (4.56 m<sup>2</sup>/sample) within 4 fields of each planting type, at 2-week intervals from 1 July to 15 August 1990 and 1991. Invertebrate abundance and biomass were lowest in early August ( $P < 0.05$ ). The CRP fields planted to a red clover/timothy mixture, and dominated by red clover, had the highest levels of invertebrate abundance and biomass ( $P < 0.05$ ). Conventionally-tilled soybeans had lower invertebrate abundance and biomass than all CRP covertypes ( $P < 0.05$ ). Mean invertebrate abundance and biomass in CRP fields were 4 times that of soybean fields. In northern Missouri, CRP fields could provide quality brood habitat if structural characteristics are also consistent with brood foraging needs. Incorporation of a legume in CRP plantings may produce higher invertebrate densities and improve the value of these fields as brood habitat.

**Burkle, L. A., C. Marlin, and T. M. Knight. 2013. Plant–pollinator interactions over 120 years: Loss of species, co-occurrence, and function. Science 339: 1611–1615.**

Using historic data sets, we quantified the degree to which global change over 120 years disrupted plant-pollinator interactions in a temperate forest understory community in Illinois, USA. We found degradation of interaction network structure and function and extirpation of 50% of bee species. Network changes can be attributed to shifts in forb and bee phenologies resulting in temporal mismatches, nonrandom species extinctions, and loss of spatial co-occurrences between extant species in modified landscapes. Quantity and quality of pollination services have declined through time. The historic network showed flexibility in response to disturbance; however, our data suggest that networks will be less resilient to future changes.

**Carvalho, L. G., Veldtman, R., Shenkute, A. G., Tesfay, G. B., Pirk, C. W. W., Donaldson, J. S. and Nicolson, S. W. (2011), Natural and within-farmland biodiversity enhances crop productivity. Ecology Letters, 14: 251–259. doi:10.1111/j.1461-0248.2010.01579.x**

Ongoing expansion of large-scale agriculture critically threatens natural habitats and the pollination services they offer. Creating patches with high plant diversity within farmland is commonly suggested as a measure to benefit pollinators. However, farmers rarely adopt such practice, instead removing naturally occurring plants (weeds). By combining pollinator exclusion experiments with analysis of honeybee behaviour and flower-visitation webs, we found that the presence of weeds allowed pollinators to persist within sunflower fields, maximizing the benefits of the remaining patches of natural habitat to productivity of this large-scale crop. Weed diversity increased flower visitor diversity, hence

ameliorating the measured negative effects of isolation from natural habitat. Although honeybees were the most abundant visitors, diversity of flower visitors enhanced honeybee movement, being the main factor influencing productivity. Conservation of natural patches combined with promoting flowering plants within crops can maximize productivity and, therefore, reduce the need for cropland expansion, contributing towards sustainable agriculture.

**Dauber, J., J. C. Biesmeijer, D. Gabriel, W. E. Kunin, E. Lamborn, B. Meyer, A. Nielsen, S. G. Potts, S. P. Roberts, and V. Söber. 2010. Effects of patchsize and density on flower visitation and seed set of wild plants: A pan-European approach. *J. Ecol.* 98: 188–196.**

Habitat fragmentation can affect pollinator and plant population structure in terms of species composition, abundance, area covered and density of flowering plants. This, in turn, may affect pollinator visitation frequency, pollen deposition, seed set and plant fitness.

2. A reduction in the *quantity* of flower visits can be coupled with a reduction in the *quality* of pollination service and hence the plants' overall reproductive success and long-term survival. Understanding the relationship between plant population size and/or isolation and pollination limitation is of fundamental importance for plant conservation.

3. We examined flower visitation and seed set of 10 different plant species from five European countries to investigate the general effects of plant populations size and density, both within (patch level) and between populations (population level), on seed set and pollination limitation.

4. We found evidence that the effects of area and density of flowering plant assemblages were generally more pronounced at the patch level than at the population level. We also found that patch and population level together influenced flower visitation and seed set, and the latter increased with increasing patch area and density, but this effect was only apparent in small populations.

5. *Synthesis.* By using an extensive pan-European data set on flower visitation and seed set we have identified a general pattern in the interplay between the attractiveness of flowering plant patches for pollinators and density dependence of flower visitation, and also a strong plant species-specific response to habitat fragmentation effects. This can guide efforts to conserve plant–pollinator interactions, ecosystem functioning and plant fitness in fragmented habitats.

**Davros, N.M., Debinski, D.M., Reeder, K.F., and Hohman, W.L., 2006, Butterflies and Continuous Conservation Reserve Program filter strips—Landscape considerations: *Wildlife Society Bulletin*, v. 34, no. 4, p. 936–943.**

Filter strips or buffers are areas of grass or other perennial herbaceous vegetation established along waterways to remove contaminants and sediments from agricultural field runoff. In the heavily cultivated regions of the Midwestern United States, these buffer zones established under the Farm Bill provide important habitat for wildlife such as butterflies. The question of how the landscape context of these plantings influences their use has not been adequately researched. We used multiple regression and Akaike's Information Criteria to determine how habitat width and several landscape-level factors (i.e., landscape composition [total herbaceous cover, amount of developed area, and amount of wooded cover] and configuration [herbaceous edge density]) influenced the abundance and diversity of the butterfly community using filter strips in southwestern Minnesota, USA. Habitat-sensitive butterfly abundance and all richness and diversity measures were positively correlated with filter-strip width. Butterfly abundance was negatively associated with the amount of developed areas (cities, towns, and roads) within the area of a 1-km radius (3.14 km<sup>2</sup>) surrounding the sites. Percentage of wooded cover in the landscape was an important variable explaining individual species abundance, although the direction of the relationship varied. Our finding that landscape context influences butterfly use of filter strips highlights the importance of landscape-level approaches to wildlife conservation in agroecosystems.

**Dormann, C.F., Fruend, J., Bluethgen, N. & Gruber B. 2009. Indices, graphs and null models: analyzing bipartite ecological networks. *The Open Ecology Journal*, 2, 7-24. Dormann, C.F. (2011). How to be a specialist? Quantifying specialisation in pollination networks. *Network Biology* 1, 1 - 20.)**

Many analyses of ecological networks in recent years have introduced new indices to describe network properties. As a consequence, tens of indices are available to address similar questions, differing in specific detail, sensitivity in detecting the property in question, and robustness with respect to network size and sampling intensity. Furthermore, some indices

merely reflect the number of species participating in a network, but not their interrelationship, requiring a null model approach. Here we introduce a new, free software calculating a large spectrum of network indices, visualizing bipartite networks and generating null models. We use this tool to explore the sensitivity of 26 network indices to network dimensions, sampling intensity and singleton observations. Based on observed data, we investigate the interrelationship of these indices, and show that they are highly correlated, and heavily influenced by network dimensions and connectance. Finally, we re-evaluate five common hypotheses about network properties, comparing 19 pollination networks with three differently complex null models: 1. The number of links per species (“degree”) follow (truncated) power law distributions. 2. Generalist pollinators interact with specialist plants, and vice versa (dependence asymmetry). 3. Ecological networks are nested. 4. Pollinators display complementarity, owing to specialization within the network. 5. Plant-pollinator networks are more robust to extinction than random networks. Our results indicate that while some hypotheses hold up against our null models, others are to a large extent understandable on the basis of network size, rather than ecological interrelationships. In particular, null model pattern of dependence asymmetry and robustness to extinction are opposite to what current network paradigms suggest. Our analysis, and the tools we provide, enables ecologists to readily contrast their findings with null model expectations for many different questions, thus separating statistical inevitability from ecological process.

**Dormann, C.F., Gruber B. & Fruend, J. (2008). Introducing the bipartite Package: Analysing Ecological Networks. R news Vol 8/2, 8 - 11.**

**Downing AS, van Nes EH, Mooij WM, Scheffer M (2012) The Resilience and Resistance of an Ecosystem to a Collapse of Diversity. PLoS ONE 7(9): e46135. <https://doi.org/10.1371/journal.pone.0046135>**

Diversity is expected to increase the resilience of ecosystems. Nevertheless, highly diverse ecosystems have collapsed, as did Lake Victoria’s ecosystem of cichlids or Caribbean coral reefs. We try to gain insight to this paradox, by analyzing a simple model of a diverse community where each competing species inflicts a small mortality pressure on an introduced predator. High diversity strengthens this feedback and prevents invasion of the introduced predator. After a gradual loss of native species, the introduced predator can escape control and the system collapses into a contrasting, invaded, low-diversity state. Importantly, we find that a diverse system that has high complementarity gains in resilience, whereas a diverse system with high functional redundancy gains in resistance. Loss of resilience can display early-warning signals of a collapse, but loss of resistance not. Our results emphasize the need for multiple approaches to studying the functioning of ecosystems, as managing an ecosystem requires understanding not only the threats it is vulnerable to but also pressures it appears resistant to.

**Ebeling, A., Klein, A.-M., Schumacher, J., Weisser, W. W. and Tschardtke, T. (2008), How does plant richness affect pollinator richness and temporal stability of flower visits? *Oikos*, 117: 1808–1815. doi:10.1111/j.1600-0706.2008.16819.x**

Pollinators play a key role in the reproduction of most plant species, and pollinator and plant diversity are often related. We studied an experimental gradient of plant species richness for a better understanding of plant–pollinator community interactions and their temporal variability, because in non-experimental field surveys plant richness is often confounded with gradients in management, soil fertility, and community composition. We observed pollinator species richness and frequency of visits six times in 73 plots over two years, and used advanced statistical analysis to account for the high number of zeroes that often occur in count data of rare species. The frequency of pollinator visits increased linearly with both the blossom cover and the number of flowering plant species, which was closely related to the total number of plant species, whereas the number of pollinator species followed a saturation curve. The presence of particularly attractive plant species was only important for the frequency of flower visits, but not to the richness of pollinators. Plant species richness, blossom cover, and the presence of attractive plant species enhanced the temporal stability in the frequency of pollinator visits.

In conclusion, grasslands with high plant diversity enhance and stabilize frequent and diverse flower visitations, which should sustain effective pollination and plant reproduction.

**Elisa Thébault, Colin Fontaine (2010) Stability of Ecological Communities and the Architecture of Mutualistic and Trophic Networks. *Science*13: 853-856**

Research on the relationship between the architecture of ecological networks and community stability has mainly focused on one type of interaction at a time, making difficult any comparison between different network types. We used a theoretical approach to show that the network architecture favoring stability fundamentally differs between trophic and mutualistic networks. A highly connected and nested architecture promotes community stability in mutualistic networks, whereas the stability of trophic networks is enhanced in compartmented and weakly connected architectures. These theoretical predictions are supported by a meta-analysis on the architecture of a large series of real pollination (mutualistic) and herbivory (trophic) networks. We conclude that strong variations in the stability of architectural patterns constrain ecological networks toward different architectures, depending on the type of interaction.

**Fleming, T. H., Sahley, C. T., Holland, J. N., Nason, J. D. and Hamrick, J. L. (2001), SONORAN DESERT COLUMNAR CACTI AND THE EVOLUTION OF GENERALIZED POLLINATION SYSTEMS. *Ecological Monographs*, 71: 511–530.**

We studied variation in flowering phenology, fruit and seed set, and the abundance of the pollinators of four species of night-blooming Sonoran Desert columnar cacti for up to eight years at one site in Mexico and one year at one site in Arizona. We determined how spatiotemporal variation in plant–pollinator interactions affects the evolution of generalized pollination systems. We conducted pollinator exclusion and hand pollination experiments to document annual variability in pollinator reliability and to determine whether pollination systems were redundant (different species are partially or totally substitutable) or complementary (different species have an additive effect on fruit set). The cacti we studied included three species with generalized pollination systems involving bats, birds, and bees (cardon, *Pachycereus pringlei*; saguaro, *Carnegiea gigantea*; and organ pipe, *Stenocereus thurberi*) and one specialized moth-pollinated species (senita, *Lophocereus schottii*). We predicted that the migratory lesser long-nosed bat, *Leptonycteris curasoae*, is a less reliable pollinator than birds and bees, and that cacti with generalized pollination systems have more variable flowering phenologies than the specialized species.

Annual time of peak flowering and mean size of flower crops were relatively invariant in saguaro and organ pipe. Time of peak flowering in cardon varied by as much as six weeks, and mean flower crop size varied three-fold over six years. In senita, peak flowering varied by as much as 5–8 wk among years. Peak numbers of the nectar bat *L. curasoae* varied among years, and bat density (0.9/ha) was an order of magnitude lower than that of cactus-visiting birds at both study sites. The abundance of migratory hummingbirds was also highly variable among years.

Pollinator exclusion experiments indicated that bats were major pollinators of cardon, whereas diurnal visitors accounted for most fruit set in saguaro (except in 1995 when bats were most important) and organ pipe at our Mexican site; honeybees accounted for 64–87% of diurnal fruit set in these species. Annual variation in the contribution to fruit set by bats was substantially higher than that of diurnal pollinators in saguaro and organ pipe, but not in cardon. There was little geographic variation in the relative importance of nocturnal vs. diurnal pollinators in saguaro and senita, but bats were much more important for fruit set in organ pipe in Arizona than in Mexico. We generally detected no effect of different pollinators on number of seeds per fruit in any species.

Annual variation in fruit set was lowest in saguaro, the species with the most diurnal pollination system, and highest in organ pipe, the species with the most generalized pollination system. Fruit set was strongly pollen limited only in females of cardon (a trioecious species) and in organ pipe (at both sites). The “missing” pollinators in both species are likely *Leptonycteris* bats. The pollination systems of saguaro and cardon were partially redundant, whereas that of organ pipe was complementary.

The four species of cactus that we studied occur at the northern geographic limits of Mexican columnar cacti where many vertebrate pollinators are seasonal migrants. In the Sonoran Desert, variation in rainfall and spring temperatures affects timing of flowering and the extent of competition between cacti for pollinator visits and causes the relative importance of particular pollinators, especially *Leptonycteris* bats, for fruit set to vary annually. Under such conditions, selection has favored generalized pollination systems (as seen in organ pipe) or shifts from reliance primarily on nocturnal pollinators (as seen in cardon) to reliance primarily on diurnal pollinators (as seen in saguaro). Nonetheless, as exemplified by the senita–senita moth system, highly specialized pollination mutualisms can also evolve in this habitat in plants that rely on sedentary insects rather than migratory bats and birds for pollination.

**FSA (2011) The Conservation Reserve Program: 39th Signup Results. Available at:**  
[https://www.fsa.usda.gov/Internet/FSA\\_File/su39book.pdf](https://www.fsa.usda.gov/Internet/FSA_File/su39book.pdf)

**FSA (2015) Conservation Reserve Program, Honey bee Health Initiative. Available at:**  
[https://www.fsa.usda.gov/Assets/USDA-FSA-Public/usdfiles/FactSheets/2015/CRPPProgramsandInitiatives/Honey\\_Bee\\_Habitat\\_Initiative.pdf](https://www.fsa.usda.gov/Assets/USDA-FSA-Public/usdfiles/FactSheets/2015/CRPPProgramsandInitiatives/Honey_Bee_Habitat_Initiative.pdf)

**Nicola Gallai N, Salles J-M, Setteled J and B E Vaissière (2009) Economic valuation of the vulnerability of world agriculture confronted with pollinator decline. Ecological Economics 68:(3) 810-821**  
<https://doi.org/10.1016/j.ecolecon.2008.06.014>

There is mounting evidence of pollinator decline all over the world and consequences in many agricultural areas could be significant. We assessed these consequences by measuring 1) the contribution of insect pollination to the world agricultural output economic value, and 2) the vulnerability of world agriculture in the face of pollinator decline. We used a bioeconomic approach, which integrated the production dependence ratio on pollinators, for the 100 crops used directly for human food worldwide as listed by FAO. The total economic value of pollination worldwide amounted to €153 billion, which represented 9.5% of the value of the world agricultural production used for human food in 2005. In terms of welfare, the consumer surplus loss was estimated between €190 and €310 billion based upon average price elasticities of - 1.5 to - 0.8, respectively. Vegetables and fruits were the leading crop categories in value of insect pollination with about €50 billion each, followed by edible oil crops, stimulants, nuts and spices. The production value of a ton of the crop categories that do not depend on insect pollination averaged €151 while that of those that are pollinator-dependent averaged €761. The vulnerability ratio was calculated for each crop category at the regional and world scales as the ratio between the economic value of pollination and the current total crop value. This ratio varied considerably among crop categories and there was a positive correlation between the rate of vulnerability to pollinators decline of a crop category and its value per production unit. Looking at the capacity to nourish the world population after pollinator loss, the production of 3 crop categories - namely fruits, vegetables, and stimulants - will clearly be below the current consumption level at the world scale and even more so for certain regions like Europe. Yet, although our valuation clearly demonstrates the economic importance of insect pollinators, it cannot be considered as a scenario since it does not take into account the strategic responses of the markets.

**Gallant AL, Euliss NH Jr, Browning Z (2014) Mapping Large-Area Landscape Suitability for Honey Bees to Assess the Influence of Land-Use Change on Sustainability of National Pollination Services. PLoS ONE 9(6): e99268.**  
<https://doi.org/10.1371/journal.pone.0099268>

Pollination is a critical ecosystem service affected by various drivers of land-use change, such as policies and programs aimed at land resources, market values for crop commodities, local land-management decisions, and shifts in climate. The United States is the world's most active market for pollination services by honey bees, and the Northern Great Plains provide the majority of bee colonies used to meet the Nation's annual pollination needs. Legislation requiring increased production of biofuel crops, increasing commodity prices for crops of little nutritional value for bees in the Northern Great Plains, and reductions in government programs aimed at promoting land conservation are converging to alter the regional landscape in ways that challenge beekeepers to provide adequate numbers of hives for national pollination services. We developed a spatially explicit model that identifies sites with the potential to support large apiaries based on local-scale land-cover requirements for honey bees. We produced maps of potential apiary locations for North Dakota, a leading producer of honey, based on land-cover maps representing (1) an annual time series compiled from existing operational products and (2) a realistic scenario of land change. We found that existing land-cover products lack sufficient local accuracy to monitor actual changes in landscape suitability for honey bees, but our model proved informative for evaluating effects on suitability under scenarios of land change. The scenario we implemented was aligned with current drivers of land-use change in the Northern Great Plains and highlighted the importance of conservation lands in landscapes intensively and extensively managed for crops.

**Gibbs J, Elle E, Bobiwash K, Haapalainen T, Isaacs R (2016) Contrasting Pollinators and Pollination in Native and Non-Native Regions of Highbush Blueberry Production. PLoS ONE 11(7): e0158937. <https://doi.org/10.1371/journal.pone.0158937>**

Highbush blueberry yields are dependent on pollination by bees, and introduction of managed honey bees is the primary strategy used for pollination of this crop. Complementary pollination services are also provided by wild bees, yetighbush blueberry is increasingly grown in regions outside its native range where wild bee communities may be less adapted to the crop and growers may still be testing appropriate honey bee stocking densities. To contrast crop pollination in native and non-native production regions, we sampled commercial 'Bluecrop' blueberry fields in British Columbia and Michigan with grower-selected honey bee stocking rates (0–39.5 hives per ha) to compare bee visitors to blueberry flowers, pollination and yield deficits, and how those vary with local- and landscape-scale factors. Observed and Chao-1 estimated species richness, as well as Shannon diversity of wild bees visiting blueberries were significantly higher in Michigan where the crop is within its native range. The regional bee communities were also significantly different, with Michigan farms having greater dissimilarity than British Columbia. Blueberry fields in British Columbia had fewer visits by honey bees than those in Michigan, irrespective of stocking rate, and they also had lower berry weights and a significant pollination deficit. In British Columbia, pollination service increased with abundance of wild bumble bees, whereas in Michigan the abundance of honey bees was the primary predictor of pollination. The proportion of semi-natural habitat at local and landscape scales was positively correlated with wild bee abundance in both regions. Wild bee abundance declined significantly with distance from natural borders in Michigan, but not in British Columbia where large-bodied bumble bees dominated the wild bee community. Our results highlight the varying dependence of crop production on different types of bees and reveal that strategies for pollination improvement in the same crop can vary greatly across production regions.

**Gould, J., 1991, Seasonal use of Conservation Reserve Program fields by white-tailed deer in eastern South Dakota: Brookings, S. Dak., South Dakota State University, M.S. thesis, 40 p.**

Seasonal use of Conservation Reserve Program (CRP) fields by white-tailed deer (*Odocoileus virginianus*) was investigated in eastern South Dakota by systematically relocating radio-marked deer from 15 September 1989 to 31 December 1990. Seasonal core areas and low use areas were determined from 40 home range maps of 14 radio-collared deer. Proportion of CRP lands within core areas of deer did not differ ( $P > 0.05$ ) from that in low use areas during each season. Pooled habitat availability within deer home ranges was similar between seasons. Habitat use varied among seasons ( $P < 0.01$ ) and was disproportional to habitat availability ( $P < 0.10$ ). Overall, CRP lands were used in greater proportions than their availability during spring, summer and fall 1990. Deer selected CRP lands during active periods in the spring and summer 1990, and during bedded periods in the summer and fall 1990. Deer avoided CRP lands during active periods in the fall 1989. Habitat selection data indicate that CRP lands provide important forage and cover values except during winter. During winter deer selected primarily wetland and wooded habitats. Partial mowing for controlling weeds, rejuvenating CRP fields, or providing emergency hay are recommended to maintain high interspersed of forage regrowth and standing cover in CRP fields.

**Goulson, D., E. Nicholls, C. Botias, and E. L. Rotheray. 2015. Bee declines driven by combined stress from parasites, pesticides, and lack of flowers. Science 347: 1255957.**

Bees are subject to numerous pressures in the modern world. The abundance and diversity of flowers has declined; bees are chronically exposed to cocktails of agrochemicals, and they are simultaneously exposed to novel parasites accidentally spread by humans. Climate change is likely to exacerbate these problems in the future. Stressors do not act in isolation; for example, pesticide exposure can impair both detoxification mechanisms and immune responses, rendering bees more susceptible to parasites. It seems certain that chronic exposure to multiple interacting stressors is driving honey bee colony losses and declines of wild pollinators, but such interactions are not addressed by current regulatory procedures, and studying these interactions experimentally poses a major challenge. In the meantime, taking steps to reduce stress on bees would seem prudent; incorporating flower-rich habitat into farmland, reducing pesticide use through adopting more sustainable farming methods, and enforcing effective quarantine measures on bee movements are all practical measures that should be adopted. Effective monitoring of wild pollinator populations is urgently needed to inform management strategies into the future.

**Grass, I., Berens D. G., and Farwig, N. 2014. Natural habitat loss and exotic plants reduce the functional diversity of flower visitors in a heterogeneous subtropical landscape. *Functional Ecology*, 28: 1117–1126.**

1. Functional diversity (FD) of pollinators can increase plant reproductive output and the stability of plant-pollinator communities. Yet, in times of world-wide pollinator declines, effects of global change on pollinator FD remain poorly understood. Loss of natural habitat and exotic plant invasions are two major drivers of global change that particularly threaten pollinator diversity.
2. In a subtropical South African landscape, we investigated changes in the FD of flower visitor assemblages on native and exotic plants along gradients of natural habitat loss and relative abundance of exotic plants. We used a data set of 1434 flower visitor individuals sampled on 131 focal plants and calculated the FD in three flower visitor traits that are strongly related to plant–flower visitor interactions and pollination processes: proboscis length, proboscis diameter and body length.
3. Multivariate FD of flower visitors decreased with both increasing natural habitat loss and relative exotic abundance. Importantly, changes in FD went beyond those in flower visitor richness. Furthermore, flower visitor richness was not related to either natural habitat loss or relative exotic abundance. Loss in multivariate FD seemed to be mediated by complementary reductions of FD in proboscis length with natural habitat loss and of FD in body length with both global change drivers. Correspondingly, we recorded lower abundances of long-tongued flower visitors with natural habitat loss and reduced variance in body size with both drivers. In contrast, FD in proboscis diameter was unaffected by either driver. All effects of the two global change drivers were non-interactive.
4. Our results show that both natural habitat loss and exotic plants negatively affect flower visitor FD, which may imperil pollination of specialized plant species in degraded habitats. In contrast, flower visitor richness may not cover all facets of flower visitor FD that are relevant to pollination processes. Distinct responses of visitor traits to the two drivers suggest limited options to infer relations of one trait to another. Finally, additive effects of natural habitat loss and exotic plant invasions highlight the need to consider multiple drivers of global change when investigating ecosystem processes at a community scale.

**Griffin, S.L., 1991, Pronghorn use of agricultural land in northwestern South Dakota: Brookings, S. Dak., South Dakota State University, M.S. thesis, 63 p.**

Use of agricultural lands by pronghorn (*Antilocapra americana*) was studied using monthly aerial and roadside surveys in northwestern Harding County, South Dakota from January 1989 to August 1990. Standing crop phytomass and crude protein content (CPC) of forages were also sampled monthly from June 1989 to July 1990 on alfalfa, small grains, native grasslands, and native sagebrush-grasslands to better understand influences of nutrient availability on habitat selection of pronghorns. Cropland, Conservation Reserve Program lands (CRP), and native prairie made up 17%, 4%, and 79% of available habitats, respectively. Pronghorn were observed foraging on cropland, CRP, and native prairie a total of 14%, 5%, and 81% of the time, respectively. Seasonal variation in the use of cropland was observed throughout the study. Use of small grains was greater than availability only during May-June 1989 ( $P \leq 0.10$ ). Alfalfa was used in proportion greater than availability during March-April and July-August in 1989, while CRP was selected in January-February 1990 ( $P \leq 0.10$ ). Alfalfa and CRP showed an inverse relationship in use by pronghorn. Mean distances that pronghorn were observed from 111 crop land did not differ from mean distances of random locations from cropland during most seasonal periods ( $P \leq 0.05$ ). Pronghorn were observed at greater distances (range 330-459 m) from roadways than were random distances ( $x = 231$  m) during all seasonal periods ( $P \leq 0.05$ ). Pronghorn selected stockponds or dugouts over creeks and the Little Missouri River and were observed at distances greater than the mean distance to water from May to August 1990 ( $P \leq 0.05$ ). Hilltops and flat areas were selected over slopes during all seasonal periods. In all vegetation types, phytomass of forage was lowest during winter dormancy, and increased during spring green-up from April through July. Phytomass of alfalfa and wheat tended to be greater than native rangelands during all times of the year. CPC levels of all forages increased as spring green-up occurred, was highest in April, and gradually decreased into the summer months. In general, pronghorn selected forage types with the greatest CPC levels during all seasons. Sport harvest together with plantings of CRP to provide alternative high-quality foraging areas are direct methods of reducing depredation on agricultural croplands.

**Hall, D. M., Camilo, G. R., Tonietto, R. K., Ollerton, J., Ahrné, K., Arduser, M., Ascher, J. S., Baldock, K. C. R., Fowler, R., Frankie, G., Goulson, D., Gunnarsson, B., Hanley, M. E., Jackson, J. I., Langellotto, G., Lowenstein, D., Minor, E. S., Philpott, S. M., Potts, S. G., Sirohi, M. H., Spevak, E. M., Stone, G. N. and Threlfall, C. G. (2017), The city as a refuge for insect pollinators. *Conservation Biology*, 31: 24–29. doi:10.1111/cobi.12840.**



Research on urban insect pollinators is changing views on the biological value and ecological importance of cities. The abundance and diversity of native bee species in urban landscapes that are absent in nearby rural lands evidence the biological value and ecological importance of cities and have implications for biodiversity conservation. Lagging behind this revised image of the city are urban conservation programs that historically have invested in education and outreach rather than programs designed to achieve high-priority species conservation results. We synthesized research on urban bee species diversity and abundance to determine how urban conservation could be repositioned to better align with new views on the ecological importance of urban landscapes. Due to insect pollinators' relatively small functional requirements—habitat range, life cycle, and nesting behavior—relative to larger mammals, we argue that pollinators put high-priority and high-impact urban conservation within reach. In a rapidly urbanizing world, transforming how environmental managers view the city can improve citizen engagement and contribute to the development of more sustainable urbanization.

**Herbertsson, L., S.A.M. Lindstrom, M. Rundlo f, R. Bommarco, and H. G. Smith. 2016. Competition between managed honey bees and wild bumblebees depends on landscape context. *Basic Appl. Ecol.* 17: 609–616.**

Honeybees might outcompete wild bees by depleting common resources, possibly more so in simplified landscapes where flower-rich habitats have been lost. We tested this by experimentally adding honeybee hives to nine sites while ensuring that ten additional sites were free from hives. The landscape surrounding each geographically separated site either held low (homogeneous landscape) or high (heterogeneous landscape) proportions of semi-natural grassland. Adding honeybees suppressed bumblebee densities in field borders and road verges in homogeneous landscapes whereas no such effect was detected in heterogeneous landscapes. The proportional abundance of bumblebee species with small foraging ranges was lower at honeybee sites than at control sites in heterogeneous landscapes, whereas bumblebee communities in homogeneous landscapes were dominated by a single species with long foraging range irrespective of if honeybees were added or not. We conclude that honeybees can impact bumblebee densities, but that landscape heterogeneity modified this effect.

**Hopwood JL (2008) The contribution of roadside grassland restorations to native bee conservation. *Biological Conservation* 141 (10):2632-2640.**

Marginal habitats such as hedgerows or roadsides become especially important for the conservation of biodiversity in highly modified landscapes. With concerns of a global pollination crisis, there is a need for improving pollinator habitat. Roadsides restored to native prairie vegetation may provide valuable habitat to bees, the most important group of pollinators. Such roadsides support a variety of pollen and nectar sources and unlike agricultural fields, are unplowed, and therefore can provide potential nesting sites for ground-nesting bees. To examine potential effects of roadside restoration, bee communities were sampled via aerial netting and pan trapping along roadside prairie restorations as well as roadsides dominated by non-native plants. Management of roadside vegetation via the planting of native species profoundly affected bee communities. Restored roadsides supported significantly greater bee abundances as well as higher species richness compared to weedy roadsides. Floral species richness, floral abundance, and percentage of bare ground were the factors that led to greater bee abundance and bee species richness along restored roadsides. Traffic and width of roadside did not significantly influence bees, suggesting that even relatively narrow verges near heavy traffic could provide valuable habitat to bees. Restored and weedy roadside bee communities were similar to the prairie remnant, but the prairie remnant was more similar in bee richness and abundance to restored roadsides. Restoring additional roadsides to native vegetation could benefit pollinator conservation efforts by improving habitat on the millions of acres of land devoted to roadsides worldwide, land that is already set aside from further development.

**Hopwood JL, Winkler L, Deal B, Chivvis M (2010) The use of roadside prairie plantings by native bees. *Living Roadway Trust Fund* [online] URL: <http://www.iowalivingroadway.com/ResearchProjects/90-00-LRTF-011.pdf> (accessed November 2011).**

Native bees such as bumble bees are important pollinators of wildflowers as well as many agricultural crops. However, recent evidence demonstrates that bumble bees are in decline in the United States. Bumble bees can utilize semi-natural areas such as gardens, hedgerows, and roadsides as habitat. We sought to better understand how bumble bees were using roadside prairie plantings by monitoring their movement within and between roadside habitats. At four sites, bumble bees were caught on roadside wildflowers, marked distinctly, and released. Sampling took place over subsequent days on

both sides of the road to determine if bees readily crossed the road. We found that nearly 800 individual bumble bees of four species were collecting either pollen or nectar from roadside wildflowers. While some bees did move linearly within a roadside to find additional flowering resources, others visited the same patch of flowers repeatedly throughout time. This strong site fidelity suggests that roadside wildflowers are a stable, consistent food source for bumble bees. In our study, bumble bees crossed roads to forage on flowers on the opposite roadside, but at much lower rates than would be expected if the road was no influence. We implemented a novel method of measuring insect mortality due to vehicles and found that no bumble bees were killed by cars, but our method may under sample larger insects. Additional research to monitor bumble bee movement between roadsides and other patches of habitat would be very worthwhile.

**Kamler, J.F., Ballard, W.B., and Swepston, D.A., 2001, Range expansion of mule deer in the Texas panhandle: The Southwestern Naturalist, v. 46, no. 3, p. 378–379.**

**Kamler, J.F., Ballard, W.B., Lemons, P.R., Gilliland, R.L., and Mote, K., 2005, Home Range and habitat use of coyotes in an area of native prairie, farmland, and CRP fields: American Midland Naturalist, v. 153, no. 2, p. 394–404.**

From 1999 to 2001 we monitored 12 coyotes (*Canis latrans*) in northwestern Texas to determine their home ranges and habitat use in a landscape interspersed with native prairie, farmland and Conservation Reserve Program (CRP) fields. Annual home range size was 10.1 km<sup>2</sup> for residents and 84.5 km<sup>2</sup> for transients. We determined habitat use at two spatial scales: within home ranges and within study area. Habitat use patterns were similar at both scales, as residents selected for native prairie and transients selected for CRP fields. Habitat use between residents and transients differed in both seasons, with residents selecting more native prairie, less farmland and less CRP (summer only) than transients. Habitat at natal den sites also differed from expected for residents, as most dens (8 of 10) were located in CRP fields. The CRP fields contained the only tall permanent vegetation on our study sites and appeared to provide important foraging habitat for transient coyotes, and denning habitat for resident coyotes.

**Kremen C, Bugg RL, Nicola N, Smith SA, Thorp RW, Williams NM (2002) Native bees, native plants, and crop pollination in California. Fremontia 30(3–4):41–49**

**Kremen C, Williams N, and R Thorp 2002. Crop pollination from native bees at risk from agricultural intensification. PNAS 99(26): 16812–16816.**

Ecosystem services are critical to human survival; in selected cases, maintaining these services provides a powerful argument for conserving biodiversity. Yet, the ecological and economic underpinnings of most services are poorly understood, impeding their conservation and management. For centuries, farmers have imported colonies of European honey bees (*Apis mellifera*) to fields and orchards for pollination services. These colonies are becoming increasingly scarce, however, because of diseases, pesticides, and other impacts. Native bee communities also provide pollination services, but the amount they provide and how this varies with land management practices are unknown. Here, we document the individual species and aggregate community contributions of native bees to crop pollination, on farms that varied both in their proximity to natural habitat and management type (organic versus conventional). On organic farms near natural habitat, we found that native bee communities could provide full pollination services even for a crop with heavy pollination requirements (e.g., watermelon, *Citrullus lanatus*), without the intervention of managed honey bees. All other farms, however, experienced greatly reduced diversity and abundance of native bees, resulting in insufficient pollination services from native bees alone. We found that diversity was essential for sustaining the service, because of year-to-year variation in community composition. Continued degradation of the agro-natural landscape will destroy this “free” service, but conservation and restoration of bee habitat are potentially viable economic alternatives for reducing dependence on managed honey bees.

**Kremen C, Williams N, Bugg RL, Fay JP, Thorp RW (2004) The area requirements of an ecosystem service: crop pollination by native bee communities in California. Ecol Lett 7:1109–1119**

Managing ecosystem services is critical to human survival, yet we do not know how large natural areas must be to support these services. We investigated how crop pollination services provided by native, unmanaged, bee communities varied on organic and conventional farms situated along a gradient of isolation from natural habitat. Pollination services from native bees were significantly, positively related to the proportion of upland natural habitat in the vicinity of farm

sites, but not to any other factor studied, including farm type, insecticide usage, field size and honeybee abundance. The scale of this relationship matched bee foraging ranges. Stability and predictability of pollination services also increased with increasing natural habitat area. This strong relationship between natural habitat area and pollination services was robust over space and time, allowing prediction of the area needed to produce a given level of pollination services by wild bees within this landscape.

**Kremen, C. N., Williams, M., and Thorp, R.W., 2006, Crop pollination from native bees at risk from agricultural intensification: Proceedings National Academy of Science, v. 99, no. 26, p. 16812–16816.**

Ecosystem services are critical to human survival; in selected cases, maintaining these services provides a powerful argument for conserving biodiversity. Yet, the ecological and economic underpinnings of most services are poorly understood, impeding their conservation and management. For centuries, farmers have imported colonies of European honey bees (*Apis mellifera*) to fields and orchards for pollination services. These colonies are becoming increasingly scarce, however, because of diseases, pesticides, and other impacts. Native bee communities also provide pollination services, but the amount they provide and how this varies with land management practices are unknown. Here, we document the individual species and aggregate community contributions of native bees to crop pollination, on farms that varied both in their proximity to natural habitat and management type (organic versus conventional). On organic farms near natural habitat, we found that native bee communities could provide full pollination services even for a crop with heavy pollination requirements (e.g., watermelon, *Citrullus lanatus*), without the intervention of managed honey bees. All other farms, however, experienced greatly reduced diversity and abundance of native bees, resulting in insufficient pollination services from native bees alone. We found that diversity was essential for sustaining the service, because of year-to-year variation in community composition. Continued degradation of the agro-natural landscape will destroy this “free” service, but conservation and restoration of bee habitat are potentially viable economic alternatives for reducing dependence on managed honey bees.

**Kutby, Rola Adel (2013) Bee Communities in Restored Landfill Sites of Niagara Region. Master’s Thesis, Faculty of Mathematica and Science, Brock University, St. Catharines, Ontario.**

This study examined the impact of habitat restoration on bee communities (Hymenoptera: Apidae) of the Niagara Region, Ontario, Canada. Bee abundance and diversity was studied in three restored landfill sites: the Glenridge Quarry Naturalization Site (GQNS) in St. Catharines, Elm Street Naturalization Site in Port Colborne, and Station Road Naturalization Site in Wainfleet during 2011 and 2012. GQNS represented older sites restored from 2001-2003. Elm and Station sites represented newly restored landfills as of 2011. These sites were compared to control sites at Brock University where bee communities are well established and again to other landfills where no stable habitat was available before restoration. The objective of this study is to investigate the impact of restoration level on bee abundance and diversity in restored landfill sites of the Niagara Region. Based on the increased disturbance hypothesis (InDH) and the intermediate disturbance hypothesis (IDH), I hypothesized that bee abundance and diversity will follow two patterns. First pattern according to InDH suggest that as the disturbance decrease the bee abundance and diversity will increased. Second pattern according to the IDH bee abundance and diversity will be the highest at the intermediate level of disturbance. A total of 7 173 bees were collected using pan traps and flower collections, from May to October 2011 and 2012. Bees were classified to five families, 21 genera and sub-genera, containing at least 78 species. In 2011 bee abundance was not significantly different among restoration levels while in 2012 bee abundance was significant difference among restoration level. According to family there were no significant difference in Halictidae and Apidae abundance among restoration level while Colletidae and Megachilidae abundance were varied among restoration levels. The bee species richness was highest in the newly restored sites followed by restored control sites, and then the control site. The current study demonstrates that habitat restoration results in rapid increases in bee abundance and diversity for newly restored sites, and, further, that it takes only 2-3 years for bee assemblages in newly restored sites to arrive at the same levels of abundance and diversity as in nearby control sites where bee communities are well established.

**Larsen KJ (2010) Impact of roadside prairie plantings on plant and insect communities LRTF Research Final Reports [online] URL: <http://www.iowalivingroadway.com/ResearchProjects/9000-LRTF-800,%20910.pdf> (accessed December 2011).**

Our goal is to monitor changes to both plant and insect roadside communities over a 5- year period as this project progresses: obtaining baseline data during 2008 (Project 90-00- LRTF-800), initial establishment during 2009 (Project 90-00-LRTF-910) after the spraying and planting during heavy weed management (primarily by mowing), and followed up by current work in 2010 (Project 90-00-LRTF-013) which is to be the second year of weed management with mowing, and then additional follow up as the planting is allowed to “go native” in 2011 and 2012 with little regular maintenance. We hypothesize that as this diverse native prairie roadside planting becomes established, there will be increased plant species richness, plant productivity, insect abundance, and insect species richness.

**Lewis, T., 1969, The diversity of insect fauna in a hedgerow and neighboring fields: Journal of Applied Ecology, v. 6, no. 3, p. 453-458.**

The insect communities on vegetation in a mixed hedgerow and in neighbouring fields of pasture and field beans were sampled with a vacuum sampler, and the aerial populations above the vegetation with suction traps. For terrestrial and aerial samples, the frequency distribution of the number of individuals in each of seventy-six taxa identified fitted a logarithmic series, so  $\alpha$  was used as an index to compare the relative diversity of communities in and above these habitats. Of the terrestrial communities, that on the hedge was the more diverse, less so in the beans, and least in the pasture. The diversity of the aerial population decreased with increasing distance from the hedge, and the resulting pattern of diversity resembled the pattern of shelter produced by the hedge. The presence of the hedge enriched the aerial population nearby for a distance of three to ten times its height to leeward and one to two times to windward.

**Losey, J.E., and Vaughan, M., 2006, The economic value of ecological services provided by insects: Bioscience, v. 56, no. 4, p. 311-323.**

In this article we focus on the vital ecological services provided by insects. We restrict our focus to services provided by “wild” insects; we do not include services from domesticated or mass-reared insect species. The four insect services for which we provide value estimates—dung burial, pest control, pollination, and wildlife nutrition—were chosen not because of their importance but because of the availability of data and an algorithm for their estimation. We base our estimations of the value of each service on projections of losses that would accrue if insects were not functioning at their current level. We estimate the annual value of these ecological services provided in the United States to be at least \$57 billion, an amount that justifies greater investment in the conservation of these services.

**McFrederick QS, LeBuhn G (2006) Are urban parks refuges for bumblebees *Bombus* (Hymenoptera: Apidae)? Biological Conservation 123:372-382.**

Declines in bee populations have been documented in several parts of the world. Bees are dependent upon flowering plants for resources, and flowering plants often depend upon bees for pollination services. Bees can therefore serve as indicator species of habitat degradation due to these relationships with flowering plants. This study investigates how the bumble bee community in San Francisco has responded to urbanization and which urban park characteristics are important for the current community's structure. To answer these questions we sampled bumble bees, in 18 urban parks and two nearby wild parks. We estimated park characteristics and used multiple regression analysis to determine which characteristics predicted bumble bee abundance and species richness. Bumble bee abundance was positively associated with resource availability or proxies of resource availability; “natural area” (areas that contain remnant fragments that have been largely unchanged by human activity) in 2003 and nest site abundance and openness of the surrounding matrix in both 2003 and 2004. Bumble bee species richness was negatively associated with abundance of a dominant species, *Bombus vosnesenskii*, in 2004. The importance of the surrounding matrix suggests that these parks do not act as islands. Accordingly, area of park did not explain species richness, while abundance of the dominant competitor did. The species that was most influenced by competition, *Bombus sitkensis*, uses rodent holes as nest sites and is possibly excluded from nest sites by the early emerging *B. vosnesenskii*, another subterranean nester. The species least influenced by competition, *Bombus melanopygus*, is able to use both rodent holes and abandoned bird nests as nest sites.

**McIntyre NE, Hostetler ME (2001) Effects of urban land use on pollinator (hymenoptera: apoidea) communities in a desert metropolis. Basic Appl Ecol 2:209-218**

We compared the species richness and abundance of pollinator (Hymenoptera: Apoidea) communities in two seasons (September 1998 and April 1999) among four types of urban land use in the Phoenix, Arizona, USA, metropolitan area (xeriscaped residential yards, mesiscaped residential yards, urban desert-remnant parks, and natural desert parks on the fringe of the metropolitan area). Richness and abundance of bees were generally lower in residential areas than in desert areas, with desert areas on the fringe of the metro area possessing the highest diversity of all sites. Residential yards that utilized xeric landscaping had a more diverse bee community (with proportionally more rare species) than did mesic (turf grass) yards, particularly in late summer (September). Although bee community structure was apparently unaffected by the *density* of local habitat features (native and exotic trees, shrubs, cacti, and herbaceous plants in addition to human-built structures), the *types* of habitat features do appear to influence the number and types of bees present in an area. These results suggest that urban development can be designed to promote the conservation of bees. Specifically, preservation of desert and greater use of xeric landscaping rather than mesiscaping may help preserve this ecologically and economically vital group of organisms.

Während zwei verschiedener Jahreszeiten (September 1998 und April 1999) verglichen wir den Artenreichtum und die Abundanz von Bienengemeinschaften zwischen vier Arten städtischer Landnutzung in Phoenix, Arizona, USA, im großstädtischen Raum (Xerophyten- und Mesophytengärten, städtische Parks mit Resten von Wüstenraum und natürliche Wüstenparks am Rand des großstädtischen Gebiets). Im Allgemeinen waren der Reichtum und die Abundanz der Bienen in Wohngegenden niedriger als in Wüstengebieten, wobei die Wüstengebieten am Rande von Stadtgebieten die größte Vielfalt von allen Plätzen besaßen. Gärten in Wohngegenden, die sich auf xerische Landschaftsbebauung verlegt hatten, spiegelten eine vielfältigere Bienengemeinschaft (mit verhältnismäßig eher selteneren Arten) wider, als solche, die mesische (Rasen)gärten angelegt hatten, und dies besonders im Spätsommer (September). Obwohl der Aufbau der Bienengemeinschaft offensichtlich unbeeinflusst blieb von der Dichte der Habitat-Merkmale (einheimische und exotische Bäume, Sträucher, Kakteen und krautartige Pflanzen zusätzlich zu von menschenhand errichteten Strukturen), scheinen die Habitattypen hingegen, den Lebensraum zu prägen und die Anzahl als auch die Art der im Gebiet angesiedelten Bienen zu beeinflussen. Diese Ergebnisse lassen vermuten, daß eine städtische Entwicklung so geplant werden kann, daß sie die Erhaltung der Bienen fördert. Insbesondere kann die Erhaltung von Wüstengebieten und eine größere Nutzung von xerischem statt mesischem Gartenbau dazu beitragen, diese ökologisch und ökonomisch lebenswichtige Gruppe von Organismen zu erhalten.

**McIntyre, N.E., and Thompson, T.R., 2003, A comparison of Conservation Reserve Program habitat plantings with respect to arthropod prey for grassland birds: *American Midland Naturalist*, v. 150, no. 2, p. 291–301.**

The Conservation Reserve Program (CRP) was designed to reduce soil erosion and curb agricultural overproduction by converting highly erodible agricultural land to various forms of perennial habitat. It has had an incidental benefit of providing habitat for wildlife and has been beneficial in reversing population declines of several grassland bird species. However, the mechanisms behind these reversals remain unknown. One such mechanism may be differences in food availability on CRP vs. non-CRP land or between different types of CRP. The influence of CRP habitat type on the abundance of arthropod prey used by grassland birds has not been previously explored. We compared the abundance and diversity of arthropods among four CRP habitat types in Texas [replicated plots of exotic lovegrass (*Eragrostis curvula*), Old World bluestem (*Bothriochloa ischaemum*), mixed native grasses with buffalograss (*Buchloë dactyloides*) and mixed native grasses without buffalograss] and native shortgrass prairie. Attention was focused on adult and juvenile spiders (Order Araneae), beetles (Coleoptera), orthopterans (Orthoptera: grasshoppers and crickets) and lepidopterans (Lepidoptera: butterflies and moths), as these taxa are the primary prey items of grassland birds during the breeding season. Arthropod diversity and abundance were higher on indigenous prairie compared to CRP, reflecting differences in vegetative diversity and structure, but there were no differences in arthropod richness or abundance among CRP types. These results indicate that, although CRP is not equivalent to native prairie in terms of vegetation or arthropod diversity, CRP lands do support arthropod prey for grassland birds. More direct assays of the survivorship and fitness of birds on CRP compared to native shortgrass prairie are clearly warranted.

**McIntyre, NE (2003) Effects of Conservation Reserve Program seeding regime on harvester ants (Pogonomyrmex), with implications for the threatened Texas horned lizard (Phrynosoma cornutum): *The Southwestern Naturalist*, v. 48, no. 2, p. 274–313.**

I compared the presence and abundance of nest-sites made by harvester ants (*Pogonomyrmex*), the primary prey for the endangered Texas horned lizard (*Phrynosoma cornutum*), among restored grassland plots planted in different grass species and indigenous prairie. The restored plots had been seeded as part of the Conservation Reserve Program (CRP) as exotic monocultures of either Old World bluestem (*Bothriochloa ischaemum*) or weeping lovegrass (*Eragrostis curvula*), or as mixtures of native grasses (both with and without buffalograss, *Buchloë dactyloides*). On average, the fewest ant mounds were found on Old World bluestem plots, whereas the indigenous grassland had the highest density of harvester ant mounds. However, there were no significant differences between native and exotic CRP plantings. Results obtained from a simultaneous visual survey for Texas horned lizards corroborate these findings. Thus, there is no evidence that CRP plots planted in exotic grasses are significantly poorer habitat for Texas horned lizards in terms of ant abundance than native grass plantings.

**Memmott J, Waser N, M Price (2004) Tolerance of pollination networks to species extinctions. Proceedings of the Royal Society B Biological Sciences. DOI: 10.1098/rspb.2004.2909.**

Mutually beneficial interactions between flowering plants and animal pollinators represent a critical 'ecosystem service' under threat of anthropogenic extinction. We explored probable patterns of extinction in two large networks of plants and flower visitors by simulating the removal of pollinators and consequent loss of the plants that depend upon them for reproduction. For each network, we removed pollinators at random, systematically from least-linked (most specialized) to most-linked (most generalized), and systematically from most- to least-linked. Plant species diversity declined most rapidly with preferential removal of the most-linked pollinators, but declines were no worse than linear. This relative tolerance to extinction derives from redundancy in pollinators per plant and from nested topology of the networks. Tolerance in pollination networks contrasts with catastrophic declines reported from standard food webs. The discrepancy may be a result of the method used: previous studies removed species from multiple trophic levels based only on their linkage, whereas our preferential removal of pollinators reflects their greater risk of extinction relative to that of plants. In both pollination networks, the most-linked pollinators were bumble-bees and some solitary bees. These animals should receive special attention in efforts to conserve temperate pollination systems.

**Morandin, L. A., and C. Kremen. 2013a. Bee preference for native versus exotic plants in restored agricultural hedgerows. Rest. Ecol. 21: 26-32.**

Habitat restoration to promote wild pollinator populations is becoming increasingly common in agricultural lands. Yet, little is known about how wild bees, globally the most important wild pollinators, use resources in restored habitats. We compared bee use of native and exotic plants in two types of restored native plant hedgerows: mature hedgerows (>10 years from establishment) designed for natural enemy enhancement and new hedgerows ( $\leq 2$  years from establishment) designed to enhance bee populations. Bees were collected from flowers using timed aerial netting and flowering plant cover was estimated by species using cover classes. At mature hedgerow sites, wild bee abundance, richness, and diversity were greater on native plants than exotic plants. At new sites, where native plants were small and had limited floral display, abundance of bees was greater on native plants than exotic plants; but, controlling for floral cover, there was no difference in bee diversity and richness between the two plant types. At both mature and new hedgerows, wild bees preferred to forage from native plants than exotic plants. Honey bees, which were from managed colonies, also preferred native plants at mature hedgerow sites but exhibited no preference at new sites. Our study shows that wild bees, and managed bees in some cases, prefer to forage on native plants in hedgerows over co-occurring weedy, exotic plants. Semi-quantitative ranking identified which native plants were most preferred. Hedgerow restoration with native plants may help enhance wild bee abundance and diversity, and maintain honey bee health, in agricultural areas.

**Morandin, L. A., and C. Kremen. 2013b. Hedgerow restoration promotes pollinator populations and exports native bees to adjacent fields. Ecol. App. 23: 829-839.**

In intensive agricultural landscapes, restoration within farms could enhance biodiversity and ecosystem services such as pollination by native pollinators. Although governments and conservation groups are promoting small-scale restoration on working farms, there are few studies that assess whether these practices enhance pollinator communities in restored areas. Further, there is no information on whether floral enhancements will deplete pollinators in adjacent fields by concentrating ambient populations or whether they result in a net increase in abundance in adjacent farm fields. We investigated whether field edges restored with native perennial plants in California's Central Valley agricultural region

increased floral abundance and potential bee nesting sites, and native bee and syrphid fly abundance and diversity, in comparison to relatively unmanaged edges. Native bees and syrphid flies collected from flowers were more abundant, species-rich, and diverse at hedgerow sites than in weedy, unmanaged edges. Abundance of bees collected passively in pan traps was negatively correlated with floral abundance, was significantly different from communities captured by net sampling from flowers, and did not distinguish between site types; we therefore focused on the results of net samples and visual observations. Uncommon species of native bees were sevenfold more abundant on hedgerow flowers than on flowers at weedy, unmanaged edges. Of the species on flowers at hedgerows, 40% were exclusive to hedgerow sites, but there were no species exclusively found on flowers at control sites. Hedgerows were especially important for supporting less-common species of native bees in our intensive agricultural landscape. Hedgerows did not concentrate ambient native bee, honey bee, or syrphid fly populations, and they acted as net exporters of native bees into adjacent fields. Within-farm habitat restoration such as hedgerow creation may be essential for enhancing native pollinator abundance and diversity, and for pollination services to adjacent crops.

**Morandin, L.A., and Winston, M.L., 2006, Pollinators provide economic incentive to preserve natural land in agroecosystems: Agriculture, Ecosystems and Environment, v. 116, p. 289–292.**

Natural habitats are considered inherently indispensable to the global economy by conservationists, but few natural ecosystems afford direct and quantifiable economic benefits. Quantification of natural land value can provide compelling evidence favoring preservation over development. Wild bees are important pollinators of many crop plants, and natural patches in agroecosystems enhance pollinator services and crop yield. Bee abundance was greatest in canola fields that had more uncultivated land within 750 m of field edges and seed set was greater in fields with higher bee abundance. A cost-benefit model that estimates profit in canola agroecosystems with different proportions of uncultivated land is presented. Yield and profit could be maximized with 30% of land uncultivated within 750 m of field edges.

**Ollerton, J., R. Winfree, and S. Tarrant. 2011. How many flowering plants are pollinated by animals? *Oikos* 120: 321–326.**

It is clear that the majority of flowering plants are pollinated by insects and other animals, with a minority utilising abiotic pollen vectors, mainly wind. However there is no accurate published calculation of the proportion of the ca 352 000 species of angiosperms that interact with pollinators. Widely cited figures range from 67% to 96% but these have not been based on firm data. We estimated the number and proportion of flowering plants that are pollinated by animals using published and unpublished community-level surveys of plant pollination systems that recorded whether each species present was pollinated by animals or wind. The proportion of animal-pollinated species rises from a mean of 78% in temperate-zone communities to 94% in tropical communities. By correcting for the latitudinal diversity trend in flowering plants, we estimate the global number and proportion of animal pollinated angiosperms as 308 006, which is 87.5% of the estimated species-level diversity of flowering plants. Given current concerns about the decline in pollinators and the possible resulting impacts on both natural communities and agricultural crops, such estimates are vital to both ecologists and policy makers. Further research is required to assess in detail the absolute dependency of these plants on their pollinators, and how this varies with latitude and community type, but there is no doubt that plant-pollinator interactions play a significant role in maintaining the functional integrity of most terrestrial ecosystems.

**Otto, C. R., L. Roth, B. L. Carlson, and M. D. Smart. 2016. Land-use change reduces habitat suitability for supporting managed honey bee colonies in the Northern Great Plains. *Proc. Natl. Acad. Sci. USA* 113: 10430–10435.**

Human reliance on insect pollination services continues to increase even as pollinator populations exhibit global declines. Increased commodity crop prices and federal subsidies for biofuel crops, such as corn and soybeans, have contributed to rapid land-use change in the US Northern Great Plains (NGP), changes that may jeopardize habitat for honey bees in a part of the country that supports >40% of the US colony stock. We investigated changes in biofuel crop production and grassland land covers surrounding ~18,000 registered commercial apiaries in North and South Dakota from 2006 to 2014. We then developed habitat selection models to identify remotely sensed land-cover and land-use features that influence apiary site selection by Dakota beekeepers. Our study demonstrates a continual increase in biofuel crops, totaling 1.2 Mha, around registered apiary locations in North and South Dakota. Such crops were avoided by commercial beekeepers when selecting apiary sites in this region. Furthermore, our analysis reveals how grasslands that beekeepers target when selecting commercial apiary locations are becoming less common in eastern North and South Dakota,

changes that may have lasting impact on pollinator conservation efforts. Our study highlights how land-use change in the NGP is altering the landscape in ways that are seemingly less conducive to beekeeping. Our models can be used to guide future conservation efforts highlighted in the US national pollinator health strategy by identifying areas that support high densities of commercial apiaries and that have exhibited significant land-use changes.

**Potts, Simon G. et al. 2010. Global pollinator declines: trends, impacts and drivers *Trends in Ecology & Evolution*, Volume 25, Issue 6, 345 – 353.**

Pollinators are a key component of global biodiversity, providing vital ecosystem services to crops and wild plants. There is clear evidence of recent declines in both wild and domesticated pollinators, and parallel declines in the plants that rely upon them. Here we describe the nature and extent of reported declines, and review the potential drivers of pollinator loss, including habitat loss and fragmentation, agrochemicals, pathogens, alien species, climate change and the interactions between them. Pollinator declines can result in loss of pollination services which have important negative ecological and economic impacts that could significantly affect the maintenance of wild plant diversity, wider ecosystem stability, crop production, food security and human welfare.

**Reeder, K.F., Debinski, D.M., and Danielson, B.J., 2005, Factors affecting butterfly use of filter strips in Midwestern USA: *Agriculture, Ecosystems and Environment*, v. 109.**

Filter strips are areas of herbaceous vegetation planted between agricultural fields and streams. In 2002 and 2003, the butterfly community in filter strips of a variety of widths and vegetative compositions was studied. Transect surveys were used to quantify butterfly abundance and diversity and measured vegetative variables in conjunction with each butterfly survey round. Overall butterfly diversity ( $H'$ ) and abundance of habitat-sensitive butterflies were positively correlated with filter strip width. Using stepwise regression, the best models to explain butterfly abundance included the coverage of forbs and the number of ramets in bloom in the strips, and indicated positive relationships between forbs and the butterfly community ( $R^2 = 0.33$  and  $0.07$ , respectively). The models that best explained abundances of large, habitat-sensitive butterflies included the height and vertical density of vegetation. The planting of forbs in filter strips is rare, but may be useful for providing food sources to butterflies.

**Ries L, Debinski DM, Wieland ML (2001) Conservation value of roadside prairie restoration to butterfly communities. *Conservation Biology* 15(2): 401-411.**

It has been suggested that restoration of roadsides to native habitat can benefit wildlife by adding habitat and restoring connectivity between fragmented reserves. In Iowa, which has one of the highest road densities in the United States, roadside vegetation has traditionally been managed to maintain a monoculture of exotic grass. Recently, several Iowa counties have begun integrated roadside vegetation management, a program that both restores roadsides to native prairie vegetation and restricts the use of herbicides and mowing. We evaluated the effect of this management regime on butterfly populations along central Iowa roadsides. We surveyed 12 separate prairie roadside areas, comparing the abundance and species richness of disturbance-tolerant and habitat-sensitive butterflies in roadside prairies with those of nearby roadsides dominated by either weeds (primarily non-native legumes) or non-native grasses. We found that management of roadsides profoundly affected the butterfly community. Species richness of habitat-sensitive butterflies showed a two-fold increase in prairie compared with grassy and weedy roadsides ( $p < 0.0001$ ), and abundance increased almost five times more on the prairie than on grassy roadsides ( $p < 0.02$ ). Species richness of disturbance-tolerant butterflies showed no response to roadside management, although abundance was slightly higher in weedy and prairie roadsides than on grassy roadsides ( $p < 0.01$ ). Of all habitat variables explored, the species richness of plants in flower showed the strongest effect on mean richness and abundance of both disturbance-tolerant and habitat-sensitive butterfly species. Although there were higher concentrations of road-killed butterflies near weedy and prairie roadsides than on grassy roadsides, relative numbers indicated that mortality risk was more than twice as high on grassy roadsides ( $p < 0.0001$ ). Tracking studies showed that butterflies were less likely to exit prairie roadsides, indicating that their mortality rates may be lower and offering preliminary evidence that roadsides have the potential to be used as corridors. Overall, our results indicate that roadside restoration benefits butterfly populations. More detailed demographic data are necessary, however, to explore the possibility that roadsides are acting as sink habitat for some habitat-restricted species.



**Ries, L., and Debinski, D.M., 2001, Butterfly responses to habitat edges in the highly fragmented prairies of central Iowa: *Journal of Animal Ecology*, v. 70, p. 840–852.**

- The behaviour of two butterfly species, a habitat specialist (*Speyeria idalia*) and a habitat generalist (*Danaus plexippus*), was tracked at four prairie edges to determine the extent to which edges act as a barrier to emigration. The four edge types studied were crop, road, field and treeline. The edges differed in structure ranging from high-contrast (treeline) to low-contrast (field).
- *S. idalia*, the habitat specialist, responded strongly to all edges, even those with low structural contrast. However, *S. idalia*'s response was strongly affected by conspecific density at crop and field edges; individuals were less likely to exit from high density plots. *S. idalia* responded to edges both by turning to avoid crossing them, and returning to the plot if they had crossed.
- *D. plexippus* responded strongly only to treeline edges. Wind direction and time of year were important factors influencing behaviour at edges for this species. Conspecific density was not a significant factor affecting their behaviour. *D. plexippus* responded to edges by not crossing them, but rarely returned once they had crossed.
- In highly fragmented landscapes, such as the one in which this study occurred, butterflies which show little or no response to edges may exhibit high emigration rates because of the high probability of encountering an edge in small habitat patches. Butterflies may respond strongly to even subtle habitat boundaries, but those responses may be modified by the edge structure, local environment or other conditions. Therefore, modifying edge structure may be a way to influence emigration rates, making it a useful tool for conservation.

**Schroeder, M.A., and Vander Haegen, W.M., 2006, Use of CRP fields by greater sage-grouse and other shrubsteppe associated wildlife in Washington: Olympia, Wash., U.S. Department of Agriculture Farm Service Agency, Washington Department of Fish and Wildlife, 39 p.**

Shrubsteppe passerines are benefiting from CRP both through creation of suitable nesting habitat and development of a more contiguous "non-cropland" landscape where CRP adjoins fragments of native shrubsteppe.

CRP was of most benefit to shrubsteppe-obligate passerines and to greater sagegrouse when it contained sagebrush and was located in a shrubsteppe landscape.

CRP is providing suitable nesting habitat for some passerine birds and for sagegrouse—those species examined were equally successful at nesting in CRP fields compared to native shrubsteppe.

CRP appears to be gaining in importance as nesting, brood-rearing, and wintering habitat for sage-grouse in Washington as the sagebrush matures.

The greater sage-grouse population in north-central Washington, an area with abundant CRP, was the only population that demonstrated an average rate of increase. This increase corresponds with the development of CRP fields into habitat with abundant sagebrush.

**Seefeldt, S.S., Conn, J.S., Zhang, M., and Kaspari, P.N., 2010, Vegetation changes in Conservation Reserve Program lands in interior Alaska: *Agriculture, Ecosystems and Environment*, v. 135, p. 119–126.**

Over 14 million hectares of erosion prone cropland in the United States has been converted into grasslands through the Conservation Reserve Program (CRP) administered by the United States Department of Agriculture, however, studies of the effects of CRP enrollment on plant communities and subsequent plant succession are largely lacking. In Delta Junction, Alaska plant communities in CRP fields are transitioning from grasslands to shrub dominated plant communities, which are resulting in compliance problems with program regulations that state "fields must be maintained in a condition that permits easy conversion to cropland". To determine plant succession and how previous land management and soils might influence the transition, we measured plant populations in 20 CRP fields throughout Delta Junction using modified-Whittaker plots. These data were combined with data on current management practices, previous farming history, soils, soil properties, diversity indices, and time since land was cleared and analyzed with nonmetric multidimensional scaling

ordination to determine factors that influence plant succession. Time in the CRP was the only factor consistently influencing plant succession. As time in the CRP increased, the planted introduced grasses brome grass (*Bromus inermis*) and red fescue (*Festuca rubra*) and the native pteridophyte (*Equisetum arvense*) decreased, whereas a native grass (*Calamagrostis canadensis*), five native forb, two native shrub, and three native tree species increased. Plant diversity increased at a rate of more than 2 species per 1000 m<sup>2</sup> per year. Regression analyses of plant species and plant groups using time in the CRP as the dependent variable resulted in the identification of outlier CRP fields with significantly more or less than expected covers of vegetation. All fields with these outliers had reasonable explanations for the differences in cover that were unrelated to the overall rate of plant succession. Current management practices will result in noncompliant fields and different management practices that result in woody vegetation control is key to maintaining CRP fields in compliance.

**Smart, M. D., S. Pettis, N. H. Euliss, Jr, and M. Spivak. 2016a. Land use in the Northern Great Plains region of the U.S. influences the survival and productivity of honey bee colonies. *Agric. Ecol. Environ.* 230: 139–149.**

The Northern Great Plains region of the US annually hosts a large portion of commercially managed U.S. honey bee colonies each summer. Changing land use patterns over the last several decades have contributed to declines in the availability of bee forage across the region, and the future sustainability of the region to support honey bee colonies is unclear. We examined the influence of varying land use on the survivorship and productivity of honey bee colonies located in six apiaries within the Northern Great Plains state of North Dakota, an area of intensive agriculture and high density of beekeeping operations. Land use surrounding the apiaries was quantified over three years, 2010–2012, and survival and productivity of honey bee colonies were determined in response to the amount of bee forage land within a 3.2-km radius of each apiary. The area of uncultivated forage land (including pasture, USDA conservation program fields, fallow land, flowering woody plants, grassland, hay land, and roadside ditches) exerted a positive impact on annual apiary survival and honey production. Taxonomic diversity of bee-collected pollen and pesticide residues contained therein varied seasonally among apiaries, but overall were not correlated to large-scale land use patterns or survival and honey production. The predominant flowering plants utilized by honey bee colonies for pollen were volunteer species present in unmanaged (for honey bees), and often ephemeral, lands; thus placing honey bee colonies in a precarious situation for acquiring forage and nutrients over the entire growing season. We discuss the implications for land management, conservation, and beekeeper site selection in the Northern Great Plains to adequately support honey bee colonies and insure long term security for pollinator-dependent crops across the entire country.

**Smart, M., J. Pettis, N. Rice, Z. Browning, and M. Spivak. 2016b. Linking measures of colony and individual honey bee health to survival among apiaries exposed to varying agricultural land use. *PLoS ONE* 11: e0152685.**

We previously characterized and quantified the influence of land use on survival and productivity of colonies positioned in six apiaries and found that colonies in apiaries surrounded by more land in uncultivated forage experienced greater annual survival, and generally more honey production. Here, detailed metrics of honey bee health were assessed over three years in colonies positioned in the same six apiaries. The colonies were located in North Dakota during the summer months and were transported to California for almond pollination every winter. Our aim was to identify relationships among measures of colony and individual bee health that impacted and predicted overwintering survival of colonies. We tested the hypothesis that colonies in apiaries surrounded by more favorable land use conditions would experience improved health. We modeled colony and individual bee health indices at a critical time point (autumn, prior to overwintering) and related them to eventual spring survival for California almond pollination. Colony measures that predicted overwintering apiary survival included the amount of pollen collected, brood production, and *Varroa destructor* mite levels. At the individual bee level, expression of *vitellogenin*, *defensin1*, and *lysozyme2* were important markers of overwinter survival. This study is a novel first step toward identifying pertinent physiological responses in honey bees that result from their positioning near varying landscape features in intensive agricultural environments.

**Tarrant S, Ollerton J, Rahman L, Tarrant J, and D McCollin (2012) Grassland Restoration on Landfill Sites in the East Midlands, United Kingdom: An Evaluation of Floral Resources and Pollinating Insects Restoration Ecology doi: 10.1111/j.1526-100X.2012.00942.x**

Pollinators are declining in Europe due to intensification of agriculture, habitat loss and fragmentation. Restored landfill sites are a significant potential reserve of semi-natural habitat, so their conservation value for supporting populations of

pollinating insects was here examined by assessing whether the plant and pollinator assemblages of restored landfill sites are comparable to reference sites of existing wildlife value. Floral characteristics of the vegetation and the species richness and abundance of flower-visiting insect assemblages were compared between nine pairs of restored landfill sites and reference sites in the East Midlands of the United Kingdom, using standardized methods over two field seasons. No differences were found between the restored landfill and reference sites in terms of species richness or abundance of plants in flower and both types of site had similar assemblages of pollinators. However, plant and insect assemblages differed across the season, with species richness and abundance being lower for the restored landfill sites in the spring and higher in the autumn compared to the reference sites. The results indicate that in this region, landfill sites are being restored to a state comparable to that of the reference sites with regards to their provision of floral resources and the associated insect pollinator assemblages. Since there are currently 2,200 working landfill sites in England and Wales, covering 28,000 ha, and closing at a rate of 100 per year, this is potentially a significant reserve of land that could be restored.

**Williams, N. M., L. Ward, N. Pope, R. Isaacs, J. Wilson, E. A. May, J. Ellis, J. Daniels, A. Pence, and K. Ullmann. 2015. Native wildflower plantings support wild bee abundance and diversity in agricultural landscapes across the United States. *Ecol. Appl.* 25: 2119–2131.**

Global trends in pollinator-dependent crops have raised awareness of the need to support managed and wild bee populations to ensure sustainable crop production. Provision of sufficient forage resources is a key element for promoting bee populations within human impacted landscapes, particularly those in agricultural lands where demand for pollination service is high and land use and management practices have reduced available flowering resources. Recent government incentives in North America and Europe support the planting of wildflowers to benefit pollinators; surprisingly, in North America there has been almost no rigorous testing of the performance of wildflower mixes, or their ability to support wild bee abundance and diversity. We tested different wildflower mixes in a spatially replicated, multiyear study in three regions of North America where production of pollinator-dependent crops is high: Florida, Michigan, and California. In each region, we quantified flowering among wildflower mixes composed of annual and perennial species, and with high and low relative diversity. We measured the abundance and species richness of wild bees, honey bees, and syrphid flies at each mix over two seasons. In each region, some but not all wildflower mixes provided significantly greater floral display area than unmanaged weedy control plots. Mixes also attracted greater abundance and richness of wild bees, although the identity of best mixes varied among regions. By partitioning floral display size from mix identity we show the importance of display size for attracting abundant and diverse wild bees. Season-long monitoring also revealed that designing mixes to provide continuous bloom throughout the growing season is critical to supporting the greatest pollinator species richness. Contrary to expectation, perennials bloomed in their first season, and complementarity in attraction of pollinators among annuals and perennials suggests that inclusion of functionally diverse species may provide the greatest benefit. Wildflower mixes may be particularly important for providing resources for some taxa, such as bumble bees, which are known to be in decline in several regions of North America. No mix consistently attained the full diversity that was planted. Further study is needed on how to achieve the desired floral display and diversity from seed mixes.

**Winfrey et al. 2007 Effect of human disturbance on bee communities in a forested ecosystem. *Conserv Biol* 21:213–223;**

It is important for conservation biologists to understand how well species persist in human-dominated ecosystems because protected areas constitute a small fraction of the Earth's surface and because anthropogenic habitats may offer more opportunities for conservation than has been previously thought. We investigated how an important functional group, pollinators (bees; Hymenoptera: Apiformes), are affected by human land use at the landscape and local scales in southern New Jersey (U.S.A.). We established 40 sites that differed in surrounding landscape cover or local habitat type and collected 2551 bees of 130 species. The natural habitat in this ecosystem is a forested, ericaceous heath. Bee abundance and species richness within forest habitat decreased, not increased, with increasing forest cover in the surrounding landscape. Similarly, bee abundance was greater in agricultural fields and suburban and urban developments than in extensive forests, and the same trend was found for species richness. Particular species groups that might be expected to show greater sensitivity to habitat loss, such as floral specialists and bees of small or large body size, did not show strong positive associations with forest habitat. Nevertheless, 18 of the 130 bee species studied were positively associated with extensive forest. One of these species is a narrow endemic that was last seen in 1939. Our

results suggest that at least in this system, moderate anthropogenic land use may be compatible with the conservation of many, but not all, bee species.

**Wright, C. K., and M. C. Wimberly. 2013. Recent land use change in the Western Corn Belt threatens grasslands and wetlands. *Proc. Natl. Acad. Sci. USA* 110: 4134–4139.**

In the US Corn Belt, a recent doubling in commodity prices has created incentives for landowners to convert grassland to corn and soybean cropping. Here, we use land cover data from the National Agricultural Statistics Service Cropland Data Layer to assess grassland conversion from 2006 to 2011 in the Western Corn Belt (WCB): five states including North Dakota, South Dakota, Nebraska, Minnesota, and Iowa. Our analysis identifies areas with elevated rates of grass-to-corn/soy conversion (1.0–5.4% annually). Across the WCB, we found a net decline in grass-dominated land cover totaling nearly 530,000 ha. With respect to agronomic attributes of lands undergoing grassland conversion, corn/soy production is expanding onto marginal lands characterized by high erosion risk and vulnerability to drought. Grassland conversion is also concentrated in close proximity to wetlands, posing a threat to waterfowl breeding in the Prairie Pothole Region. Longer-term land cover trends from North Dakota and Iowa indicate that recent grassland conversion represents a persistent shift in land use rather than short-term variability in crop rotation patterns. Our results show that the WCB is rapidly moving down a pathway of increased corn and soybean cultivation. As a result, the window of opportunity for realizing the benefits of a biofuel industry based on perennial bioenergy crops, rather than corn ethanol and soy biodiesel, may be closing in the WCB.

**Vaudo, A.D., Tooker, J.F., Grozinger, C.M. et al. 2015. Bee nutrition and floral resource restoration. *Current Opinion in Insect Science*, 10: 133-141.**

Bee-population declines are linked to nutritional shortages caused by land-use intensification, which reduces diversity and abundance of host-plant species. Bees require nectar and pollen floral resources that provide necessary carbohydrates, proteins, lipids, and micronutrients for survival, reproduction, and resilience to stress. However, nectar and pollen nutritional quality varies widely among host-plant species, which in turn influences how bees forage to obtain their nutritionally appropriate diets. Unfortunately, we know little about the nutritional requirements of different bee species. Research must be conducted on bee species nutritional needs and host-plant species resource quality to develop diverse and nutritionally balanced plant communities. Restoring appropriate suites of plant species to landscapes can support diverse bee species populations and their associated pollination ecosystem services.

Appendix 1: Enrollments by county in the study states (2013), noting number of acres and number of PHP enrollments. Counties where research was conducted are indicated in bold.

IOWA			NEBRASKA			MONTANA		
COUNTY	NUMBER	ACRES	COUNTY	NUMBER	ACRES	COUNTY	NUMBER	ACRES
ADAIR	23	75	ADAMS	3	14	BIG HORN	3	71
ADAMS	6	42	ANTELOPE	19	48	BLAINE	5	79
ALLAMAKEE	1	0	BOONE	9	64	CASCADE	2	20
AUDUBON	4	20	BOYD	3	22	CHOUTEAU	13	226
BENTON	2	3	BUFFALO	4	21	DANIELS	1	4
BREMER	1	3	BURT	2	4	FERGUS	9	107
BUCHANAN	2	2	BUTLER	7	22	GARFIELD	10	327
BUTLER	15	36	CASS	4	8	HILL	4	68
CALHOUN	12	18	CEDAR	13	48	JEFFERSON	1	38
CARROLL	2	7	CHASE	7	38	JUDITH BASIN	2	14
CASS	13	64	CHERRY	1	13	MCCONE	12	24
CERRO GORDO	1	1	CHEYENNE	15	223	PETROLEUM	5	30
CHEROKEE	1	1	CLAY	6	16	PHILLIPS	6	207
CHICKASAW	4	10	CUMING	3	3	PONDERA	4	28
CLARKE	11	72	CUSTER	10	54	PRAIRIE	1	3
CLAY	1	10	DAWSON	3	18	STILLWATER	4	49
CLAYTON	9	35	DEUEL	2	5	<b>TETON</b>	<b>13</b>	<b>57</b>
CLINTON	1	2	DODGE	4	7	TOOLE	15	127
CRAWFORD	8	11	DUNDY	3	12	VALLEY	2	29
DALLAS	1	2	FILLMORE	1	3	WIBAUX	1	13
DECATUR	10	87	<b>FRANKLIN</b>	<b>26</b>	<b>137</b>	YELLOWSTONE	1	7
DICKINSON	4	14	FRONTIER	2	1	STATE	114	1720
DUBUQUE	26	123	FURNAS	66	263			
FAYETTE	3	9	GAGE	3	22			
FLOYD	7	31	GARDEN	4	15			
FRANKLIN	1	1	GARFIELD	1	18			
FREMONT	1	4	GOSPER	1	1			
GREENE	15	23	GREELEY	10	66			
GUTHRIE	2	5	HALL	1	1			
HARDIN	2	6	HARLAN	8	33			
HENRY	12	36	HAYES	3	42			
HOWARD	1	4	HITCHCOCK	1	4			
IDA	1	6	HOLT	20	140			
JACKSON	24	98	HOWARD	12	52			
JASPER	1	6	JEFFERSON	11	38			
JEFFERSON	5	19	JOHNSON	2	9			
JONES	7	19	KEITH	6	48			
KOSSUTH	1	10	KEYA PAHA	1	17			
LEE	8	21	KIMBALL	15	339			
LINN	2	12	KNOX	27	124			
LUCAS	27	113	LANCASTER	4	7			
MADISON	43	175	LINCOLN	6	61			
MAHASKA	1	1	LOGAN	1	3			
MARION	4	17	MCPHERSON	1	29			
MARSHALL	1	1	MORRILL	4	6			
MONONA	2	6	NANCE	6	33			
MONROE	1	8	NEMAHA	10	17			
MONTGOMERY	2	9	NUCKOLLS	5	25			
MUSCATINE	1	3	OTOE	16	44			
OSCEOLA	4	6	PAWNEE	9	23			
PALO ALTO	1	2	PERKINS	2	29			
PLYMOUTH	1	1	PIERCE	3	21			
POLK	2	8	POLK	1	1			
E POTTAWATTAMIE	3	8	RED WILLOW	2	11			
POWESHIEK	10	22	RICHARDSON	4	9			
RINGGOLD	8	53	ROCK	2	32			
SAC	10	29	SAUNDERS	5	16			
STORY	1	1	SCOTTS BLUFF	2	5			
TAYLOR	3	17	SEWARD	1	2			
UNION	1	6	SHERIDAN	19	246			
VAN BUREN	15	38	SHERMAN	9	61			
WAPELLO	1	7	THAYER	9	51			
WARREN	2	7	THURSTON	2	3			
WASHINGTON	12	39	VALLEY	7	26			
WAYNE	27	141	WAYNE	2	3			
WEBSTER	1	3	WEBSTER	12	81			
WINNEBAGO	1	1	WHEELER	8	204			
<b>WOODBURY</b>	<b>10</b>	<b>41</b>	STATE	491	3062			
WORTH	5	22						
WRIGHT	2	1						
STATE	453	1734						