NATIVE POLLINATORS: THE EFFECTS OF LIVESTOCK GRAZING ON MONTANA RANGELANDS

by

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ABSTRACT

Although native pollinators on rangelands serve, in part, as food at higher trophic levels, their primary ecosystem function is pollination. With 70% of western U.S. rangelands grazed by livestock, understanding how grazing affects native pollinators is a key component to managing rangelands, yet it is not well understood. In this study, I investigated how cattle grazing influences both diversity and abundance of native pollinators, including bees and wasps, syrphid flies, butterflies, and moths at two research locations for six to 10 weeks during the spring of 2016, 2017, and 2018. The first site, near Sidney, MT, served to understand if pollinators were more closely associated with active cattle grazing or rested pastures. The second site, near Roundup, MT, served to understand if pollinators were more associated with pastures either enrolled or not enrolled in the Sage Grouse Initiative, or pastures that had not experienced livestock grazing in previous seven years. Colored pan traps were deployed weekly in each treatment at each site. In addition to pollinator collections, weekly measurements of vegetation via Daubenmire frame were also collected. I collected 17,078 specimens at Sidney and 13,683 specimens at Roundup. My results suggest that in drier sagebrush landscapes, native pollinators are positively to neutrally associated with pastures moderately grazed by livestock. However, in mixed grass prairie landscapes that receive high precipitation, rest-rotational grazing does not appear to have a significant effect on primary native pollinators.

CHAPTER ONE

INTRODUCTION AND LITERATURE REVIEW

Background of Rangeland Grazing in the Western United States and Montana

There are more than 307 million hectares of rangelands in the United States (Havstad *et al*., 2009; USDA, 2018) and approximately 80% of these lands lie west of the 95th meridian (Derner & Jin, 2012), which, in the United States, extends from Minnesota south to the eastern edge of Texas. In the western U.S., these diversified rangeland landscapes consist of a wide variety of forbs, grasses, and shrubs (Kauffman & Pyke, 2001) that, in addition to serving as a critical food source for wild and domesticated ungulates, also serve as a vital microhabitat for arthropod communities such as detritivores, recyclers, and pollinators (Cock *et al*., 2012). Approximately 4,000 species of arthropod pollinators (Michener, 2007) take to the fields each growing season to collect pollen and nectar necessary to sustain themselves during fall and winter months. This mutualistic relationship (Bronstein, 1994; Harmon *et al*., 2011) accounts for roughly 35% of crop-animal pollination (Klein *et al*., 2007) and is intertwined with higher trophic levels that continue to be threatened by agricultural intensification and habitat conversion (Cunningham, 2000; Kremen *et al*., 2007; Winfree *et al*., 2009). Although, 70% of western rangelands are managed with livestock grazing (Krausman *et al.,* 2009), the interactions between livestock and pollinators is not well understood (Yoshihara *et al*., 2008).

In 2018, the United States was the largest beef producer worldwide (USDA ERS, 2018) with approximately 94.4 million head of cattle (USDA NASS, 2018), a standing that has relatively low variability year to year. The presence of livestock grazing on rangelands lying west of the 95th meridian has the potential to alter these lands through the duration, timing, and intensity of grazing (Vallentine, 1990; Rinella & Hileman, 2009; Black *et al*., 2011). However, livestock grazing is neither inherently positive or negative and continues as the most predominant global use of grasslands and rangelands (Coppock *et al*., 2009). As such, grazing and herbivory are dominant pressures on rangeland plant communities and thus, influence the quality or state of a vegetation area (DiTomasso *et al*., 2017). The ways in which rangeland landscapes can be altered or converted from a predominant state include climate, geomorphology, topography, and external disturbances like herbivory or fire (Black *et al*., 2011). Herbivory may influence vegetation communities in positive or negative ways by changing vegetation composition, productivity, seed dispersal, and succession patterns. This consequently impacts higher trophic levels like arthropod communities (Rambo & Faeth, 1999).

In order to understand how herbivory may impact vegetation, it is first important to understand how cattle graze. For example, cattle naturally have preferences toward new green growth on grasses, forbs, and woody plants (Lyons & Machen, 2000). When new green growth is not present, they shift grazing toward relatively older green growth and stems, and finally toward dry vegetative biomass (Lyons & Machen, 2000). Additionally, cattle graze on average 7-12 hours per day; however, time spent feeding is highly dependent on forage availability and stocking rate such that a high stocking rate equates to more time spent grazing with less forage available per animal (Lyons &

Machen, 2000). In addition to stocking rate, topography, distance to water, and supplementation of cattle also influences how and where cattle graze (Vallentine, 1990). Cattle tend to congregate and spend more hours per day at a given watering site, in areas with flat or rolling landscapes, and open and/or shaded areas. These preferences tend to leave pastures unevenly grazed or patch grazed (Lyons & Machen, 2000), which can alter vegetation composition and stage of succession/degradation (Rambo & Faeth, 1999; Vulliamy *et al*., 2006) as well as arthropod communities (Saarinen & Jantunen, 2005). Cattle have also been documented to directly trample potential or occupied nest sites of arthropods known to be critical rangeland pollinators (Vulliamy *et al*., 2006). Furthermore, rangelands that are routinely over-grazed display patches of native plant cover which can allow for non-native invasive plants to establish and often out-compete native plants for soil and water resources (Cane & Neff, 2011). One widely studied and problematic example is the cheatgrass *(Bromus tectorum)* invasion of western North America. Though cheatgrass was introduced by humans in the late 1800's (USDA, 2008), cattle grazing may disperse seeds and reduce native competitors allowing cheatgrass to further establish (Diamond, 2009); however, fire is also a major contributor to the expansion of cheatgrass in the western U.S. (Balk *et al*., 2013). Additionally, Rocky Mountain juniper (*Juniperus scopulorum Sarg*) and Utah juniper (*Juniperus osteosperma* [Torr.] Little) invasion of grazed Idaho shrublands is partly thought to be accelerated by cattle grazing and removal of competitors (Sankey & Germino, 2008). One 2008 study conducted near Pocatello, Idaho used satellite imagery to conclude that juniper abundance was greater in areas routinely grazed by cattle (Sankey & Germino, 2008). Additionally, Sankey and Germino (2008) suggest the results of their analysis are consistent with literature that suggests livestock grazing in shrublands facilitates juniper encroachment as it promotes soil moisture and ground cover through the increase of shrubs that assist in the establishment of safe sites for juniper growth (Gottfried, 1992). However, while Sankey and Germino (2008) analyzed juniper encroachment from an area where livestock grazing was held relatively constant for a 20-year period in areas with forage availability ranging between 0.01-0.24 Animal Unit Months (AUMs) per hectare, invasive species establishment or encroachment may vary in differing stocking rates, AUMs, or even grazing management (Milchunas & Vandever, 2014). Invasive species can have detrimental impacts on biodiversity at many levels, often influencing the natural plant-insect interactions (Traveset & Richardson, 2006; Bartomeus *et al*., 2008). For example, by altering rangeland flora and introducing more wind-pollinated plants such as cheatgrass, native pollinators are pushed out of critical nesting or foraging sites as encroachment of alien species reduces the composition and availability of floral, nectar, and soil reproductive resources (Cock *et al.*, 2012; Stout & Tiedeken, 2017).

In addition, as the early 1900's settlers flocked to the United States for homesteading, rangeland management and livestock production became an empirically studied niche (Holecheck, 1981). As a result, various livestock grazing systems (i.e., seasonal, continuous, and rest-rotation grazing) were implemented and studied in order to properly manage range and pasture lands (Holecheck, 1983). Seasonal grazing is the process of cattle grazing during the growing season for about 2-6 months, depending on the region, while continuous grazing is defined as livestock grazing during an entire 12-month period including both the dormant and growing seasons (Holecheck, 1983). Rest-rotational grazing, developed in the 1950's by a United States Forest Serviceman Gus

Hormay (Holecheck, 1983), is defined by resting a pasture from grazing for a minimum of one year, while other pastures are grazed separately based on season of use. Each grazing type has its strengths and weaknesses and ultimately the best grazing system is determined by the intended use of the land (Hanselka *et al*., 2012). Interestingly, rest-rotation cattle grazing is most often used for conservation purposes and land restoration programs (Sayre *et al*., 2012; Enri *et al*., 2017; Lyons *et al*., 2017). Rest-rotation livestock grazing is designed to provide plants a recovery and regenerative period from herbivory (Budd & Thorpe, 2009); in turn providing opportunity to increase physiological resistance or build up root reserves at some capacity, to cattle grazing (Briske *et al*., 2011). For example, a study of the influence of prairie vegetation cover on grouse lekking behavior concluded that the second highest density of nests and broods for sharp-tailed grouse (*Tympanuchus phasjanellur*) and prairie chickens (*Tympanuchus* spp.) were located in lands managed with rest-rotation grazing, in contrast to winter only grazing (Rice & Carter, 1982). Furthermore, a 1984 study of elk forage and livestock conflicts found that rest-rotational grazing enhanced vegetation in rested pastures in ways that sustained both elk and forage, while still maintaining economic viability for ranchers (Grover & Thompson, 1986; Krausman *et al*., 2009).

Insects within grazed-managed areas tend to show a range of responses to plant architecture when compared to unmanaged areas (Gibson *et al*., 1992). In addition, there is evidence that rest-rotation grazing can increase arthropod diversity (Gibson *et al*., 1992) which may be beneficial to pollinating insects if a heterogeneous landscape of ample flowering resources was maintained (Enri *et al*., 2017). Herbivory may also contribute to patch dynamics and create vegetative diversity within various early and late successional plant communities (Rambo & Faeth, 1999), thus allowing for a wide diversity of arthropods to coexist, if in fact arthropod diversity is a direct result of plant species and structural diversity. The relatively large populations of arthropods and quick generation turnover also uniquely place species like rangeland pollinators as important bioindicators of rangeland health and the impact, such as livestock, have on these landscapes (García *et al*., 2009). For example, if an area is grazed too intensively, this may reduce the abundance and diversity of arthropods (Gibson *et al*., 1992) thus reducing species richness by limiting floral or vegetation resources (Black *et al*., 2011). In contrast, when properly managed through grazing timing and intensity (Vallentine, 1990; Rinella & Hileman, 2009; Black *et al*., 2011), livestock use can increase plant biomass and in turn, native bee diversity as both are inherently linked (Vulliamy *et al*., 2006; Enri *et al*., 2017). This ecological pathway supports the concept of functional diversity through the intermediate disturbance hypothesis where plant and arthropod diversity would be highest with moderate levels of disturbance (Rambo & Faeth, 1999; Vulliamy *et al*., 2006; Yan *et al*., 2015). It also provides the framework necessary for the habitat heterogeneity hypothesis where increased plant diversity provides more arthropod niches which facilitate a sound coexistence of arthropods (Lyons *et al*., 2017) when competitive and dispersal species coexist (Roxburgh *et al*., 2004; Shea *et al*., 2004; Barnes *et al*., 2006).

The Importance of Rangelands

Rural Economy. Worldwide, approximately 1 billion people rely on rangeland biomes as a source of food, water, fiber, and income (MA, 2006; FAO, 2009; Sayre *et al*., 2013). Native rangelands are the backbone to the livelihoods of many rural families and communities. However, as the world population grows, so does the monetary value of undeveloped rangelands for conversion to either croplands or urbanization. Approximately half of the United States' western rangelands are publicly owned and leased to private cattle ranching operations (Cameron *et al*., 2014). In order to sustain an operation that is largely less profitable on a per-unit-basis, in an era that may leave agricultural landscapes depleted of quality soils, landowners have turned to diversifying their operations by working with ecological and/or conservation organizations to replace the negative footprint of agriculture with positive conservation outcomes (Behnke, 2008; Sayre *et al*., 2013). Working with conservation organizations allow ranchers options of federal tax breaks, alternative incomes, assistance in fixing ranch infrastructures like fences and watering troughs, and land restoration programs (Sayre *et al*., 2013) which help to maintain and improve habitat quality necessary for native and wildlife as a whole (de Steiguer *et al*., 2008).

The concept of incentive-based land management was first introduced as the Farm Bill by President Roosevelt in 1933. The intended purpose of the original Farm Bill was to better manage and govern various aspects of food and agricultural production (Winders, 2018). The Farm Bill is updated every 5-years, with the most recent version published in 2014, and accounts for thirteen core themes including nutrition, crop insurance, and conservation. Particularly relevant to native pollinator conservation is Title II which advocates for the preservation of land and the use of conservation easements, protections, stewardships, and partnerships with landowners. Within Montana, the Natural Resources Conservation Service (NRCS), a branch within the

United States Department of Agriculture (USDA), works to assist landowners who implement Farm Bill conservation programs on their land (NRCS, 2014). Some of these programs give special consideration to the preservation and inclusion of pollinator friendly habitat on privately owned lands through the EQIP (Environmental Quality Incentive Program), Conservation Reserve Program (CRP), and the Conservation Reserve-Enhancement Program (CREP) (USDA, 2015). The attention of this program has made available $4 million in funds for six northern states, (Montana, Michigan, Minnesota, North Dakota, South Dakota, and Wisconsin) where two-thirds of the U.S. honeybees are located during the summer foraging season (Fritscher, 2015). Many of the Farm Bill programs can and do have positive effects on native pollinator habitat, but the implementation of additional assistance in these six states would contribute to even healthier landscapes for native pollinator conservation. Although the Farm Bill can be at times controversial, it is, in general, designed to work with producers while providing many forms of assistance to U.S. citizens (Johnson & Monke, 2013).

Environment. Western U.S. rangelands do serve, in part, to the livelihoods of a global population; however, the ecosystem services they provide facilitates many trophic relationships (Beschta *et al*., 2013). Some of these services include: 1) provisioning: products we obtain from the environment like fiber and water, 2) regulating: benefits we obtain from natural processes like water filtration or carbon sequestration, or 3) supporting processes: the presence of biodiversity to assist in other ecosystem services (Yahdjian *et al*., 2015) which provide immeasurable economic value that could be threatened through a degradation of native rangeland productivity (Herrick *et al*., 2013).

However, as one of the most biodiverse ecosystems in the world, rangeland provisioning services cater to a wide suite of biodiversity (Swinton *et al*., 2007).

One of the most notable regulating services rangelands provide is that of carbon sequestration (Sala *et al*., 2017). Carbon sequestration, as defined by the United States Geological Survey, is the process of which atmospheric carbon dioxide is captured and stored (USGS, 2018). Within rangelands, carbon sequestration occurs primarily in the soils, which accounts for the largest quantity of carbon sequestrated and for the most variability in the interannual world carbon sink (Ahlström *et al*., 2015). Geographically, most of the United States' rangelands lay west if the 95th Meridian in the Rocky Mountain and great plains regions and account for the largest capacity for long-term carbon sequestration (USGS, 2018).

Western U.S. rangelands encompass many ecosystems and biomes including sagebrush and grass dominated prairies. The sagebrush *Genus Artemisia* is a dominate shrub in much of western North America (McArthur, 2000). However, sagebrush ecosystems have been reduced in recent decades from a historical 100-million ha to less than 43-million hectares (Rowland, 2006). This reduction is a direct result of human facilitated change and botanical encroachment of non-native and invasive species (Davies *et al*., 2011) that together exacerbate changes in plant communities and their resilience to invasion (Reisner, 2010). Along with non-native invasion comes with the potential of increased fire fuel load and continuity which leads to a decrease in the fire return interval, further favoring the establishment of exotic invasive grasses (Davis *et al*., 2011). It is estimated that various sagebrush steppe ecosystems require 30 –240 years to return to pre-fire conditions (Davies, 2012) which is impossible when exotic grass dominated systems establish fire return intervals of less than a decade. In addition to the described disturbances above, changes in climatic conditions, such as temperature, precipitation, humidity, also influence ecosystem functioning (Welch, 2005). As various outlets predict winter and summer rainfalls to change within Montana, we can anticipate even more changes in native rangeland plant communities in addition to human and alien plant induced disturbances (Welch, 2005). Anthropogenic disturbances like herbicide treatments, in addition to domestic livestock grazing, influence the vegetative biodiversity within sage dominated ecosystems (Mueggler, 1980), consequently contributing to the contraction of Greater Sage Grouse birds in the west by 56% (Schroeder *et al.*, 2004), and by one third within Montana alone (Peters, 2017).

Montana is home to twenty-seven species of sagebrush, that range from woody to non-woody, and inhabit diverse ecoregions from mesic to xeric (Wambolt, 2004). In neighboring Great Basin states, sagebrush has been found to occur in denser patches on more xeric landscapes, whereas mesic landscapes have been found to produce interspersed patches of sagebrush (West, 1988). Similarly, within western Montana ecosystems, big sagebrush plants are generally found with various species of forbs, legumes, and herbs (Mueggler, 1980) in soils varying in pH 5.9 up to pH 10 (Welch, 2005). Sagebrush generally serves as the dominant over story in some Montana landscapes, with grasses and forbs serving as critical understory (Pennington *et al*., 2016).

One important avian species that occupies sagebrush steppe habitats is the Greater Sage Grouse (*Centrocercus urophasianus*) (hereafter: GSG) which has become an important ecological indicator linked to native rangeland health. However, populations of sage grouse rapidly dropped as sagebrush steppe habitats began to decrease in size. As a result, the USDA NRCS works regionally to assist individuals with the Farm Bill program procedures and the Sage Grouse Initiative (hereafter; SGI) was implemented in 2010 to protect and monitor birds in native rangelands within Montana (USDA, 2014). While GSG populations were assessed for Federal listing in 2015 under the Endangered Species Act by the Fish and Wildlife Service (FWS), it was ultimately decided not to list GSG in response to an 11-state alliance, including private and public partners, to preserve 90% of GSG breeding habitat (FWS, 2017). Now at more than 1100 partners (Montana Sage Grouse Habitat Conservation Program, 2017), NRCS is working to improve 1.78 million hectares of GSG habitat, with NRCS dedicating another $211 million and aiming to protect approximately 3.2 million additional hectares (Sage Grouse Initiative, 2015). This initiative, although intended for the protection of GSG birds, cascades through the ecosystem and serves as an indicator for invertebrate and forb productivity (Rowland, 2006). As a part of this integral GSG coalition, the state of Montana has initiated its own means of preservation via the 'Montana Sage Grouse Habitat Conservation Program'. Habitat depletion and alteration, in addition to invasive species contribute to the degradation of critical sagebrush steppe ecosystems that cater to GSG population stability. As a result, the Montana Sage Grouse Habitat Conservation Program has established a review process for projects, recreation, or activities that overlap with NRCS established GSG core areas (Montana Sage Grouse Habitat Program 2017). This process serves as an integral aspect of better land management practices that contribute to the protection of sagebrush steppe ecosystems and consequently the conservation of sagebrush obligate species.

Additionally, prairies or grasslands are important rangeland and grazing ecosystems. Historically size of all these rangelands is estimated at approximately 10,359,952 hectares of North American land (NPS, 2018) and has been significantly reduced in modern decades by agricultural intensification and urbanization (Samson & Knopf, 1996; Fuhlendorf *et al*., 2002). Despite this, North American prairies still provide vital habitat, food, and water resources for a diversity of fauna including birds, mammals, and insects (USDA USFS, Accessed 2019). Native pollinators are critical in maintaining many native plant populations on these prairies where habitat fragmentation may limit pollen production as suitable habitats become smaller pockets distributed on the landscape (Kearns *et al*., 1998; Moncada, 2003). One study by Öckinger *et al*. (2006) assessed how semi-natural grasslands impact bumble bee and butterfly populations concluding that intact habitats in agricultural areas leads to increased diversity and abundance while habitat fragmentation reduced these metrics. In addition to providing critical resources to a suite of organisms, prairies are home to various prairie grouse. Prairie grouse, much like GSG, are an important umbrella species to prairie ecosystems (Poiani *et al*., 2001; Pruett *et al*., 2009). The sharp tailed grouse (*Tympanuchus phasianellus* L.) and its subspecies are also products of their ecosystems in that declining populations are often a result of overgrazing of cattle, altered fire regimes, and agricultural intensification (NRCS, 2007).

Pollinators as a Critical Component in Montana Ecosystems

Degraded native habitats have a detrimental effect to the native flora and fauna. Pollinators require pollen and nectar for optimal survivorship (Pywell *et al*., 2006; Klein *et al*., 2007; Isaacs *et al*., 2009). Conservation and revegetation of rangelands does assist in landscape rehabilitation; however, revegetation with native flowering plants ultimately provides the highest relative abundance of native pollinators. The flowering periods of native plants overlap with native pollinator needs and ultimately provides more resources over the prolonged summer months where pollinators are actively gathering resources for reproduction and winter hibernation (Fiedler & Landis, 2007; Tuell *et al*., in press; Isaacs *et al*., 2009). Additionally, Tuell *et al*. (in press), found that the diversity and abundance of native bees over the summer growing season improved with revegetation using native plant species when compared to non-native vegetation. Availability of native floral resources in an ecosystem not only influences which pollinator species are present on the landscape but this native vegetation community also positively increases native bee richness and abundance (Banaszak, 1996; Steffan-Dewenter & Tscharnke, 2001; Potts *et al*., 2003).

The presence of available plant resources is largely affected by spatial, temporal, and pollination mediated changes (Kremen *et al*., 2007). In turn, this largely impacts what plants, grasses, forbs, and shrubs may be present on the landscape. While grasses self-pollinate, forbs require cross-pollination by animals, primarily arthropods. This is a process where an insect, (i.e., a bee) transfers pollen from the stamen of one flower to the stigma of another. Forbs, an important habitat component, are commonly found on western sagebrush steppe and prairie landscapes. One importance of forbs is nitrogen fixation of legume species within the soil by capturing atmospheric nitrogen and converting it to free usable nitrogen in the form of soil ammonia, nitrite, or nitrate (Editors of the Encyclopedia of Britannica, Accessed 2019).

In addition to nitrogen fixation, rangeland forbs are also important staples of grouse diets and harbor arthropods on which grouse also feed (Johnsgard, 1983; Savory, 1989; Moss, 1997) and provide shelter and cover against predation and environmental elements (Pennington *et al*., 2016). While arthropods are important to post hatch grouse diets, transition from a primarily arthropod rich diet to an herbaceous forb and arthropod diet is essential in grouse biology (Blomberg *et al*., 2013); making forbs and arthropod diversity essential in overlapping rangelands.

Just as important as forbs, sagebrush landscapes are also home to some 350 additional facultative and obligate species that rely on these landscapes throughout the course of a full or partial year (Connelly *et al.*, 2004). In particular, big sagebrush (*Artemisia tridentata*), is known to be a key source of forage for wildlife in winter seasons as snow blankets bury forb species (Welch, 2005). In one such case, Crawford *et al*. (2004) describe the importance of sagebrush in adult sage grouse as being as important as arthropods are to chick diets. Additionally, a study conducted by Wiens *et al*. (1991) found Oregon sagebrush landscapes support as many as 76 different species of insects. Notably, a review of big sagebrush communities by Welch *et al*. (2005) reports that many arthropod species, including more than 120 species, are associated with sagebrush landscapes.

Pollinators as Critical Components to Rangelands

Hymenoptera. As of 2011, the class Insecta made up 81.6 % of the worlds' arthropod biomass (ASU, 2012). Of this, Hymenoptera (i.e., bees, wasps, and ants) ranks second in total diversity with over 150,000 described species (Huber, 2009). With approximately 15,000 new insect species being documented on an annual basis (Nosowitz, 2015), there are now approximately >20,000 known species of bees worldwide; 4,000 of which reside within the United States alone (Gilgert & Vaughan, 2011; Hoffman, 2011). As the second most diverse class of Insecta, Hymenopterans occupy multiple biomes as they inhabit every continent on Earth, except Antarctica (Wilson-Rich, 2014).

The ancestors of native pollinating bees were first documented in conjunction with the evolution of plant blossoming; colorful and sweet-scented flowers arose in the fossil record about 120-million years ago (Grimaldi & Engel, 2005; Wilson-Rich, 2014). However, it is their rapid radiation and split from parasitizing wasps that has allowed pollinating bees to truly thrive on the landscape. The superfamily Apoidea, comprised of anthophila and spheciform wasps, differentiates bees from wasps through many anatomical characteristics of which include branched setae, sub-antennal sulci, and even a meso tibial comb that is utilized for scraping pollen from scopa (Grimaldi & Engel, 2005). In essence, native bees are holometabolous, vegetarian wasps (Grimaldi & Engel, 2005) and their presence are dependent on both the availability of and diversity of floral resources (Levy, 2011).

As primary pollinators, female non-parasitizing bees (native or honey), actively collect pollen on the pollen collecting hairs, or scopa, on either their hind legs, or the underside of their abdomen. As to provide reward or act as a lure, many flowers evolved to overcompensate pollen production for fertilization, thus allowing excess to fall off the bees' body and onto the ovules of other flowers (Michener, 2000). In addition to pollen collection as food, female bees collect pollen and nectar to provision and rear their young. Both eusocial and solitary bees collect pollen resources: eusocial bees (like the non-native European honeybees) fill cells within some structure such as a honeycomb, while solitary (native bees) store pollen with freshly deposited eggs within their brood chambers (Michener, 2000). Most bees obtain the ability to repeatedly sting predators with a modified ovipositor on the end of its abdomen, serving as its only defense mechanism. Although honeybees have popularized the concept of bees living within hives, approximately 70% of native bees nest within hollowed out soil cavities while the remaining 30% use cavities in decaying vegetation (Black *et al*., 2011). Some bees like the mason bee (*Osmia spp*.) or leaf cutting bee (*Megachile spp.*) typically nest in a tube-like cavity that is filled or compartmentalized by mud or circular pieces of leaves or flower petals (Wilson & Carril, 2015). As the eggs eclose (hatch), the larvae eat the stores of pollen that their mother has left for them and chew their way through the cavity in which they inhabit (Wilson & Carril, 2015). Other bees, like *Calliopsis* of the family Andrenidae, secrete wax-like substances to line or waterproof their soil born nest in order to keep water from annual floods out of their nest (Wilson & Carril, 2015). Each bee species is unique in many ways, adapting various techniques to survive and reproduce in wet, dry, or extreme climates, some of which still exhibit a range of sociality dependent on climate and resource availability (Michener, 2007).

Lepidoptera. Lepidoptera, butterflies and moths, are a relatively new lineage only radiating in the fossil record about 100 million years ago (New, 2013). This order consists of approximately 47 super families (Kristensen *et al*., 2007), divided into three informal groupings: butterflies, macro-moths, and micromoths (New, 2013). While each are unique, in general, Lepidopterans are a holometabolous insect; developing into distinct intervals of egg, larva, pupa, and adult. Larva are a soft bodied insect with hardened mouth parts (Triplehorn & Johnson, 2005) and tend to occupy distinctly different habitats (New, 2013). After building up enough fat stores, the immature Lepidopteran forms a protective casing called a cocoon and finally pupates into an adult that will live for up to two weeks. The adults are widely known for their charismatic colorful patterns exhibited on their wings, expressed through tiny scales. While these patterns are attractive to look at, they also serve an important role in mating and defense (Meyer, 2016). Adults are also unique when compared to the immature by developing morphological characteristics such as the tympana organ. This organ is a vibrating structure within adults that is thought to be used in predator detection, similar to the concept of echo-locating calls of bats (Triplehorn & Johnson, 2005). Most adults feed on nectar resources through a proboscis that assists in lapping or sucking the resource (Triplehorn & Johnson, 2005), most larva feed on green, dead, or dying biomass. Additionally, Lepidopteran larvae are one the most important and intensively studied crop pests (Mathews, 1999; New, 2013). The silkworm (*Bombyx mori*) is a large crop and forest tree pest (Meyer, 2016) which can act as agents of succession within plant communities when local abundances are sufficient (Culin, 2018). Monocultured croplands are very susceptible to defoliation and destruction by Lepidopteran pests. Heterogeneity across a landscape, including crop fields, may help diversify food preferences for these insects mitigating the transition of food preferences.

Lepidopterans serve as an important yet secondary pollinator on rangelands (Black *et al*., 2011). Within North America, there are approximately 4,500 identified

Lepidopterans that visit floral resources regularly (Travers *et al*., 2011). While there are always exceptions, most Lepidopterans are generalists, feeding from white, pink, and red tubular flowers with strong floral scents. A variety of species are diurnal, active day and night, and have the ability to contribute to pollination services throughout the day. However, Lepidopterans lack the advanced pollen carrying structures that are unique to their Hymenopteran counterparts. Lepidopteran roles in pollination come as a side effect of nectar consumption.

While grasslands are one of the most pronounced ecosystems worldwide, many species of Lepidopterans rely on grassland ecosystems and populations are easily impacted by grazing regime in a variety of countries (Littlewood, 2008; Rickert *et al*., 2012). Grazing cattle have direct impacts on these insects through manure spread and trampling (New, 2013). However, Lepidopterans appear to be secondarily impacted by cattle grazing through the expression of available plant and nectar resources (Runquist, 2011). Some available studies have demonstrated the impacts of cattle grazing, management, and intensity on endangered or threatened Lepidopteran species such as the Large blue (*Maculinea arion*) (Thomas, 1991), and IUCN Red-Listed butterflies (Balmer & Erhardt, 2000).

Diptera. Dipterans comprise approximately 12% of insect biomass (Grimaldi & Engel, 2005), with ~7,000 documented species (Chandler, 2017). While this order is quite common in many parts of the world, often mimicking other insects, Dipterans are distinguished by the presence of halteres. Halteres are knobbed structures that have replaced hind wings and assist the insect in flying and hovering (Triplehorn & Johnson, 2005). Dipterans are usually soft-bodied, holometabolous insects and are commonly important vectors of disease to both humans and animals. This insect has either spongy sucking mouthparts or piercing mouthparts like mosquitos. In addition to their metamorphosis biology from a maggot (larvae) to adult, the larvae of this order may live in extreme environments like the *Helaeomyia petrolei* (Coquillett) that lives in bodies of crude petroleum (Triplehorn & Johnson, 2005).

Many Dipterans serve quite important roles within the field of agriculture; attacking noxious weeds or even contributing to plant pollination (Triplehorn & Johnson, 2005). While the plant-insect relationships between Dipterans and pollination is not as widely studied as that of Hymenopteran pollinators, it has been documented that Dipterans are frequent and regular visitors to at least 555 plant species (Larson *et al*., 2001). Many species of Dipterans are generalist feeders; however, it is thought that Syrphid flies (hover flies or flower flies) are an important group of flowering plant pollinating insects (Larson *et al*., 2001). Some flies have been documented to consume pollen granules and even feed off of floral nectar supplies (Haslett, 1989a; Haslett, 1989b; Irvin *et al*., 1999; Larson *et al*., 2001; Woodcock *et al*., 2014) indirectly contributing to plant pollination through a ‘mess and soil’ relationship (Faegri & van der Pijl, 1979). Additionally, Syrphidae foraging activity on flowers varies through time of day, peaking early in the morning to gain most floral rewards (Faegri & van der Pijl, 1979) and when it can warm its flight muscles through sun basking (Inouye *et al*., 2015). This group of flies are distinguished from others in a morphological characteristic present within their wings called a spurious vein (Sengupta, 2016). Flies, including the Syrphidae, do not contain pollen carrying structures, nor are their short mouth parts evolved to extract pollen and nectar from deep flowers (Kearns & Inouye, 1994). Therefore, much of their foraging are limited to open, bowl shaped flowers (Kearns & Inouye, 1994).

Modern Role in Pollination. In 2014, the Obama Presidential Administration initiated a nation-wide mandate to assess pollinator health in response to the increasingly dangerous colony collapse disorder in domesticated honeybees (Pollinator Health Task Force, 2015). As a result, this memorandum set aside 5 goals, one of which includes setting aside 2.83 million hectares to increase wild pollinator habitat (Pollinator Health Task Force, 2015). This memorandum successfully popularized and initiated a nation-wide evaluation of wild and domesticated pollinator health through academic, government, and civilian research.

Today, about $235-$577 billion of annual global food supplies rely on pollination services (FAO, 2016). Of this evaluation, 85% is achieved through insects in which are responsible for 80% of crop pollination (Allsop *et al*., 2008) and native bees alone account for 80% of flowering plant pollination (Moisset & Buchmann, 2011). In less than a decade, this plant-insect interaction has become one of the world's most valuable regulatory ecosystem services (Kremen *et al*., 2007; Klein *et al.*, 2007). However, with agricultural intensification and habitat conversion, pollination by both wild and non-wild bees is becoming increasingly threatened and the services provided by pollinators much more valued on a global scale (Koh *et al*., 2016). Interestingly, since the beginning of colony collapse disorder in domesticated honey bees, farmers and apiarists have begun to collect, farm, and mildly domesticate wild bees, such as the alfalfa leaf cutting bee

(*Megachile rotundata*), or mason bee (*Osmia lignaria*) in order to compensate for losses associated with honey bee health declines.

Rangeland pollinators are vital to and important in the functioning of many biomes. In addition to providing a direct benefit to flowering plant species seeking pollination services, these arthropods provide other animals, such as songbirds, turkey, sage grouse, and sharp tailed grouse with a necessary food source. Gilgert and Vaughan (2011) indicate that 61% of birds that breed within the United States are insectivorous and another 25% are partially insectivorous. As such, pollinators also serve as a key resource in the survival and enrichment of bird species within the U.S. Native pollinators also cater to secondary sources as their services cascade through trophic level in the form of seeds or fruits which provide megafauna such as grizzly bears, coyotes, and elk with food sources (Gilgert & Vaughan, 2011). Consequently, the effects of pollination on overall rangeland health can be measured in many and varied ways in that these landscapes are systematically structured on the services native pollinators provide.

CHAPTER TWO

EFFECTS OF LIVESTOCK GRAZING ON POLLINATORS- SIDNEY

Introduction

Pollination by wild (native) and managed bees is a critical ecological and economic service in the U.S. (Kremen *et al*., 2007). While catering to many ecosystems, pollinators inevitably face pesticide exposure, habitat reduction, disease, and reduced floral resources, among many other pressures, that all contribute to their decline (Cane & Tepedino, 2001). The continued loss of pollinators expands beyond direct effects, especially since they deliver an ecosystem service that without many plant species would simply cease to exist (Kevan & Collins, 1974; Kearns *et al*., 1998). However, heterogeneous and semi-natural grasslands which comprise many areas of Montana landscape offer promising refuge for the conservation of pollinating species within agricultural landscapes as they provide a diverse suite of resources, that include pollen and nectar resources, as well as nesting sites (Öckinger & Smith, 2007; Delaney *et al*., 2015).

Pollinators are an inherent part of a healthy range landscape (Black *et al*., 2011). Central to many prey, nutrient cycling, and food production webs within rangelands, pollinators are essential to energy-rich food webs (Black *et al*., 2011; Kearns *et al*., 1998). However, much of the western United States is managed with cattle grazing of which eastern Montana is no exception, and therefore investigations into the relationship between native pollinators and livestock grazing is necessary. As a part of a Montana

Fish, Wildlife and Parks (here after MT FWP) conservation easement, three privately-owned adjacent ranches within the northern great plains near Sidney, Montana entered with an emphasis on a rest-rotational grazing system for the conservation of sharp-tailed grouse. I further investigated the influence of the rotational grazing system on the pollinators of Hymenopteran, Dipteran, and Lepidopteran in spring to early summer for three consecutive field seasons. I hypothesize pastures that are rested during the spring season will be beneficial to wild pollinators.

Study Area

The northern great plains have a rich ecological history, coevolving with heavy bison grazing for approximately 10,000 years (Mack & Thompson, 1982). This short and mixed grass ecosystem spans 74 million hectares across five states, including Montana (Haggerty *et al*., 2018; WWF, Accessed 2018) and has been identified as one of the most threatened U.S. biomes (Hoekstra *et al*., 2005; Olimb & Robinson, 2019). However, almost 200 years ago, the northern great plains were dominated by buffalo, elk, grouse, and now-endangered black-footed ferret (WWF, Accessed 2018). In the late 1800's the great plains were significantly disturbed and deteriorated with European settlement (Wang *et al., 2014*) and the Dust Bowl era in the early 1900's (Wang *et al., 2016*). As one of the largest uses for uncultivated grasslands worldwide (Gillson & Hoffman, 2007), large animal grazing impacts many aspects of an ecosystem such as above ground net primary productivity (Li *et al*., 2005; Rogers *et al*., 2005) through plant consumption and dung and urine deposits (McNaughton *et al*., 1988). Additionally, the northern great plains are a point of important ecological monitoring for many private, state, and/or federal entities including the World Wildlife Fund (WWF), the United States Department of Agriculture (USDA), and state Land Grant Universities. Of the approximately 74 million hectares, approximately half is privately owned and utilized for cattle or beef production, but also provide critical conservation areas for threatened or endangered species (Haggerty *et al*., 2018). Conservation of habitats and provision of rangeland ecosystems services can be achieved through science based and managed livestock grazing practices (Brunson & Huntsinger, 2008).

Methods

Pollinator Surveys

Assessment of bee diversity and abundance is often measured with malaise traps, vane traps (McCravy & Ruholl, 2017), sweep-netting, and pan traps (Popic *et al.*, 2013) in a variety of ecosystems. However, the pan trap method is most commonly used for pollinator studies as it is cost effective and a time efficient passive sampling method (Leong & Thorp, 2001; Westphal *et al*., 2008) with low collector bias (Saunders & Luck, 2013). Common items used as pan traps are plastic deli bowls, pans, or cups that are filled with a mixture of soapy water (Droege, 2012) or a 50:50 mixture of propylene glycol and water (Packer, 2012) to act as a killing agent and/or preservation agent. Both mixtures decrease surface tension, generally drown a landing insect within a minute, and have the potential to collect insect by-catch (Droege, 2015). One study recorded that 33% of bees escape pan traps regardless of the drowning solution (Droege, 2015). Pan traps which are colored inside and out may be used to mimic the visual appearance of floral resources to enhance bee visitation rates (Saunders & Luck, 2013). Blue, white, and yellow pan traps are most commonly used (Leong & Thorp, 2001) and align well with floral resources of our Sidney location.

Weather and Degree Days

Wind, temperature, and precipitation averages were retrieved from the Sidney-Richland KSDY METAR weather station located at 47.7000 and -104.2000 (USpest.org). While 30-year precipitation averages were not available for the Sidney-Richland KSDY METAR weather station, averages were found on the NOAA Climate Data online web page from the SDY KSDY weather station located at 47.7283 and -104.147 (US Climate Data, 2019). Degree days were calculated by selecting the Sidney Richland KSDY METAR weather station and setting lower threshold to 0˚C and the upper to 30˚C utilizing the single sine method (USpest.org).

Setting Traps

Our methods followed Droege (2012) who outlined bee bowl sampling. As outlined in Figure 1 below, 10 to 14 pastures were sampled yearly at the Sidney location; including fall and spring grazed pastures and rested pastures all three years of study. Winter grazed and off-easement pastures were included in years 2016 and 2018. Details highlighting pasture size, turn-in and out dates, *etc.*, can be found in Table 1 below. In 2016 and 2017 and in the approximate center of each pasture (i.e., experimental unit), a single transect of pollinator traps was randomly deployed within each pasture. Each transect contained a total of nine plastic cups (16 oz. Solo®, Lake Forest, IL): three blue, three yellow, and three white extending in a straight line, approximately three meters apart (Droege *et al.*, 2010). For sampling years 2016 and 2017, each trap was set on the ground in shallow hole leaving 90% of the colored trap exposed. Trap holes were generated with a standard one-piece soil auger (AMS®, American Falls, ID), traps were placed approximately 2.5 cm deep in the hole and then backfilled with remaining soil to provide stability for the cup. In order to keep traps highly visible, vegetation was flattened down and away from traps during the growing season. Traps were replenished weekly from a one-gallon plastic jug filled with tap water and one tablespoon of unscented dish soap (Dawn®, P&G, Cincinnati, OH) where each trap was filled approximately ¾ full. Additional eight-gallon water containers and dish soap were stored in the truck in order to refill jugs for each transect.

In 2018 and in the approximated center of each pasture, nine traps were mounted on three stands where each stand held one yellow, one blue, and one white trap. Each stand was constructed of one piece of 2.5 cm diameter pvc pipe cut to 91 cm in length and held vertically by sliding the pvc over a 1.5 cm diameter piece of rebar driven into the ground with a hammer. Three 15.24 cm diameter pvc rings cut approximately 7.62 cm thick were connected to each vertical pvc stand using two black UV-resistant zip-ties (Gardner Bender, Menomonee Falls, WI) in a fashion so that cup traps would fit inside the ring and be held off the ground. To protect each trap stand, a 71.12 cm tall by 152.4 cm wide graduated mesh metal wire fencing (Garden Zone, Charleston, SC) was then placed to encircle each individual trap and a 5-inch piece of wire was used to taper the top half of the fencing shut. The fencing and tapering technique was utilized to reduce cattle disturbance to traps.

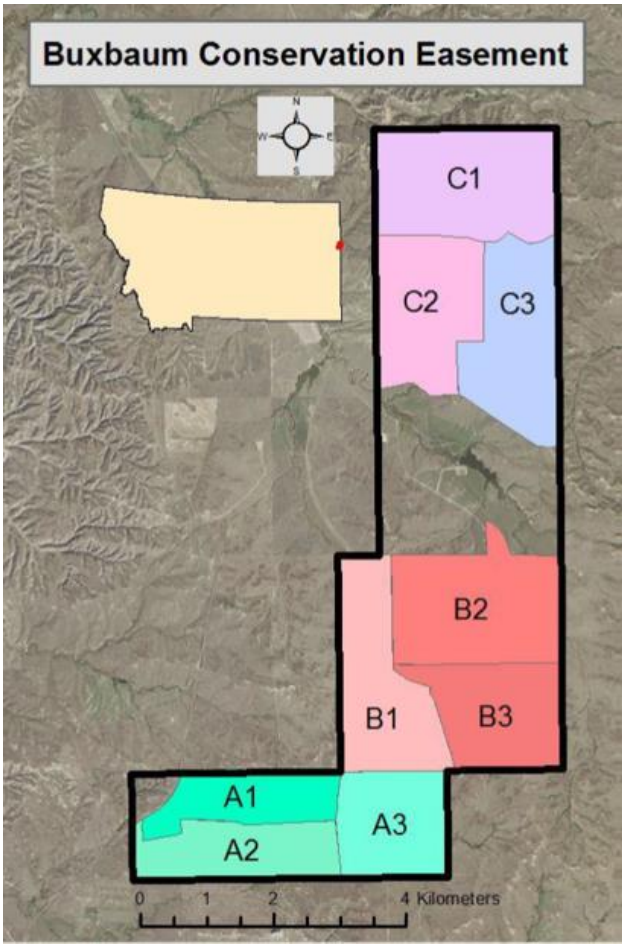

Figure 1. Study area in Richland County, MT during 2016, 2017, and 2018 sampling season. Transects of pollinator pan traps within rest-rotational grazing complex with pastures denoted with a letter number combination in order to indicate its grazing management.

Table 1. Pastures within Sidney, MT study, year, pasture size (ha), number of cattle heads, turn dates, number of grazed days, and cow days per ha.

Pasture	Year	Pasture size (ha)	Number of head	Turn-in date	Turn-out date	Number of days grazed	Cow days/ha
A1	2016	211	171	15-Aug	17-Oct	61	49.49
A2	2016	256	0	Rested	Rested	0	0
A3	2016	263	171	15-June	15-Aug	61	39.71
B1	2016	364	0	Rested	Rested	0	0
B2	2016	434	221	14-June	1-Aug	47	23.93
B3	2016	310	221	1-Aug	15-Nov	107	76.28
C1	2016	453	195	1-Aug	15-Nov	107	46.06
C2	2016	346	0	Rested	Rested	0	0
C3	2016	371	105	14-June	1-Aug	47	13.30
A1	2017	211	0	Rested	Rested	0	0
A2	2017	256	45	20-June	9-Aug	50	8.79
A3	2017	263	45	9-Aug	1-Oct	53	9.07
B1	2017	364	176	5-June	8-Aug	64	30.95
B2	2017	434	170	8-Aug	5-Nov	89	34.86
B3	2017	310	0	Rested	Rested	0	0
C1	2017	453	0	Rested	Rested	0	0
C2	2017	346	130	5-June	8-Aug	64	24.05
C3	2017	371	130	8-Aug	5-Nov	89	31.19
A1	2018	211	0	Rested	Rested	0	0
A2	2018	256	0	Rested	Rested	0	0
A3	2018	263	34	10-June	15-Nov	158	20.43
B1	2018	364	160	4-Aug	15-Nov	103	45.27
B2	2018	434	0	Rested	Rested	0	0
B3	2018	310	160	5-June	4-Aug	60	30.96
C1	2018	453	160	5-June	4-Aug	60	21.19
C2	2018	346	160	4-Aug	15-Nov	103	47.63
C3	2018	371	0	Rested	Rested	0	0

Collecting

As denoted above, traps were deployed seven days per week and the contents were always collected and traps reset on the same day. Replicate catches from the three similar colored traps were combined in Whirl-Paks® (Nasco, Fort Atkinson, WI) pre-labeled with cup color, date, and pasture locality. Large non-pollinator debris, such as rocks, sticks, leaves, etc. were removed from each sample on location. Whirl-Paks® were then stored as is in a cooler. Traps were visually inspected for cracks or damage and re-filled ¾ full of soapy water and replaced in the same color pattern and transect as initially determined.

Processing Samples

Whirl-Paks® were returned to the Montana State University laboratory where samples were cleaned. Guidelines for cleaning and processing samples are detailed in *The Handy Bee Manual* (Droege, 2015) and modified as necessary. My process was transferring Whirl-Pak® contents to a fine mesh soil strainer, removing additional large debris, rinsing with cool water, and finally replacing in separately marked Whirl-Paks® with enough 95% ethanol solution to cover and preserve the specimens. At a later date, preserved specimens were transferred to a petri dish where members of Lepidoptera and Syrphidae were identified and counted. Hymenopteran specimens containing branched hairs were pulled from samples, labeled, and kept in a petri dish in a freezer (-17 ℃) until further processing could occur. Branched hairs bees were placed in a 473 mL Ball® mason jar (Ball Corporation, Broomfield, CO) with one drop of unscented dish soap (Dawn®, P&G, Cincinnati, OH) and approximately 29 mL of water. The mason jar top was secured, and the contents shaken for about 60 seconds to remove additional small debris, clean matted hairs, and generally prepare the specimens for identification. Cleaned specimens were transferred to a paper towel and excess water blotted away. After about 3-minutes, bees were transferred to a dry mason jar containing three small balled-up pieces of paper towel (to assist in soaking up excess water and aid in fluffing the fine hairs on the bee bodies) and the jar opening was secured with a rubber band and fine mesh net top. The contents were dried by shaking the jar while forcing warm air over the specimens using a commercial hair dryer (Conair®, Conair Corporation, Stamford, CT). After this drying process, specimens were pinned, labeled, and stored in airtight specimen drawers until identified. Specimens were identified to *Genus* utilizing *The Bee Genera of North and Central America* key (Michener *et al*. 1994) and individuals were recorded by color, date, and location within an excel spreadsheet.

Vegetation Cover

Vegetation cover, frequency, and composition were measured weekly upon collection of pollinator traps as established by Daubenmire (1959). Working at the beginning of a pollinator trap transect, a 20 cm x 50 cm frame was placed at five intervals three meters apart extending in a single direction away from the transect. Using the quadrat markings on the frame, vegetation was denoted as either forb, flowering forb, grass, litter, shrub, or bare ground on a scale of 1-6 indicating an approximate percentage range of coverage and finally the midpoint of range. Table 2 provides the range of coverage associated with each cover class (Daubenmire, 1959).

Table 2. Daubenmire cover class range of coverage percentages.

Cover Class	Range of Coverage	Midpoint of Range
1	0-5%	2.5%
2	5-25%	15.0%
3	25-50%	37.5%
4	50-75%	62.5%
5	75-95%	85.0%
6	95-100%	97.5%

Statistical Analysis

Pollinator count data and Daubenmire vegetative cover data were analyzed within sampling year and combined over study years. All data were fit to a generalized linear mixed model with sampling location as the random variable (SAS, 2002). Akaike information criterion (AIC) values were used to assess the relative quality of candidate model distribution where smaller AIC values indicate a better model fit and subsequent explanation of the data. This process resulted in pollinator count data being fit to a negative binomial distribution and Daubenmire vegetative cover data being fit to a normal distribution within the generalized linear mixed model. Differences among treatment least squared means were calculated using the least significant difference test.

Results

We collected a total of 17, 074 pollinator specimens over the course of three collecting seasons from 2016 to 2018 at the Sidney, Montana field location. With a total of 19 sampling weeks spanning from May to July over three years, we collected a total of 26 Hymenoptera *Genera* from a rest-rotation grazing system and pastures that are either fall or spring grazed, or rested with sampling seasons 2016 and 2018 including both winter grazed and off-easement pastures.

Activity-Density

In 2016 at the Sidney site, treatment least squared means of weekly activity-density, a function of (pollinator activity across a landscape) x (population density) (Hokkanen & Holopainen 1986) did not differ for total pollinators and total Hymenoptera (Table 3). Secondary pollinator activity-density was greatest in the winter pastures when compared to fall ($t = 5.61$; df = 4, 10; $p < 0.01$), rested ($t = 4.95$; df = 4, 10; $p < 0.01$), and off-easement pastures ($t = 5.46$; df = 4, 10; $p < 0.01$). Secondary pollinator activity-density ranked second in spring pastures when compared to fall ($t = 2.31$; df = 4, 10; $p = 0.02$), rested ($t = 2.86$; df = 4, 10; $p = 0.04$), or off-easement pastures ($t = 2.54$; df = 4, 10; $p = 0.01$). Fall, rested, and off-easement pastures did not differ (Table 3, Fig. 2).

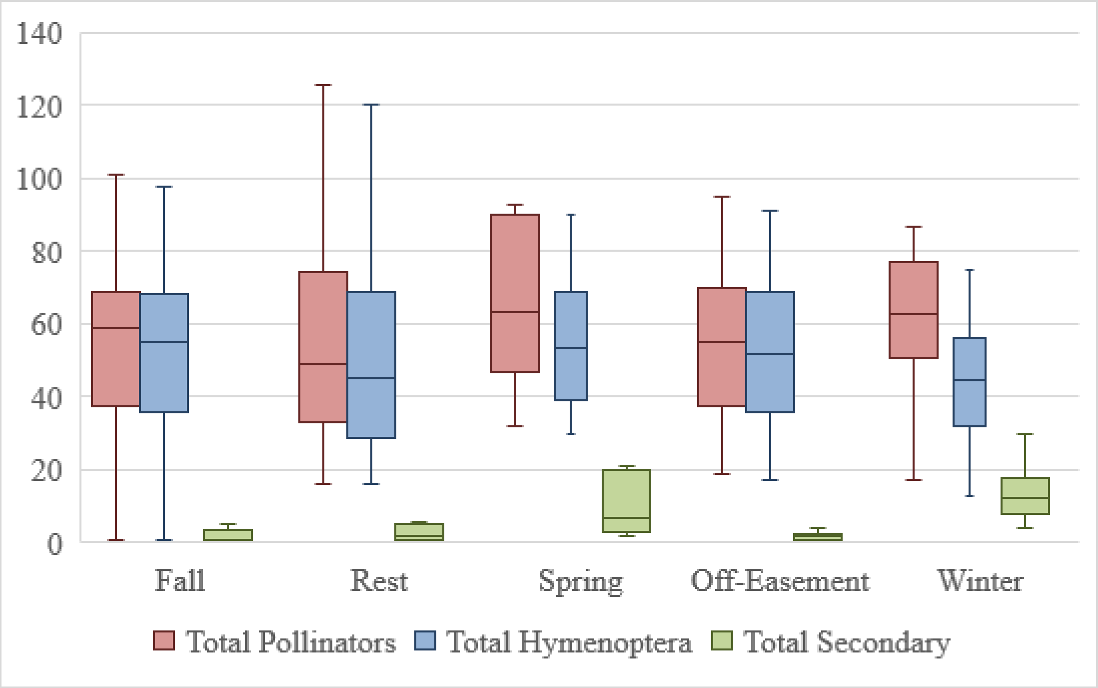

Figure 2. Boxplots of weekly total, Hymenoptera, and secondary pollinator activity-density collected from the 2016 field season east of Sidney, MT. The top and bottom whiskers indicate the largest and smallest observations within 1.5 interquartile ranges, the box represents the first and third quartile ranges, and the median is indicated by the horizontal line.

In 2017 at the Sidney site, least squared means of treatment level activity-density did not differ in total pollinators, Hymenoptera, and secondary pollinators (Table 3, Fig. 3).

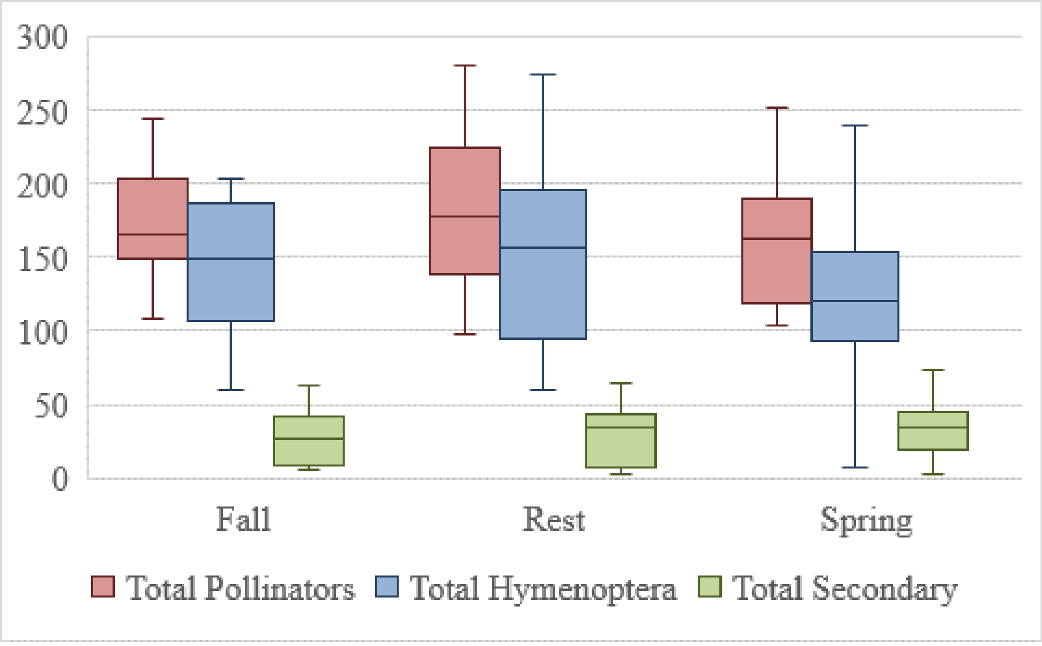

Figure 3. Boxplots of weekly total, Hymenoptera, and secondary pollinator activity-density collected from the 2017 field season east of Sidney, MT. The top and bottom whiskers indicate the largest and smallest observations within 1.5 interquartile ranges, the box represents the first and third quartile ranges, and the median is indicated by the horizontal line.

In 2018 at the Sidney site, least squared means of treatment level activity-density for total pollinators was greatest in the off-easement pastures compared to rest (t = 2.18; df = 4, 10; p = 0.05), fall (t = 2.18; df = 4, 10; p = 0.05), and winter (t = 4.37; df = 4, 10; $p < 0.01$) pastures. Total pollinator activity-density in spring pastures did not differ from off-easement, rest, or fall pastures but was greater (t = 3.13; df = 4, 10; $p < 0.01$). Hymenoptera activity-density was greatest in off-easement pastures when compared to fall (t = 2.18; df = 4, 10; p = 0.04), rest (t = 2.04; df = 4, 10; p = 0.05), spring (t = 2.04; df = 4, 10; p = 0.05), and winter (t = 4.29; df = 4, 10; $p < 0.01$) pastures. Hymenoptera activity-density was also greatest in the fall (t = 2.61; df = 4, 10; p = 0.01), rest (t = 2.73; df = 4, 10; $p < 0.01$), and spring (t = 2.73; df = 4, 10; $p < 0.01$) pastures when compared to winter pastures; which did not differ from each other. Secondary pollinator activity-density was greatest in off-easement (*t* = 1.99; df = 4, 10; *p* = 0.05) and spring (*t* = 2.30; df = 4, 10; *p* = 0.03) pastures when compared to winter pastures. Secondary pollinator activity-density in rest and fall pastures did not differ from spring, off-easement, and winter pastures (Table 3, Fig. 4).

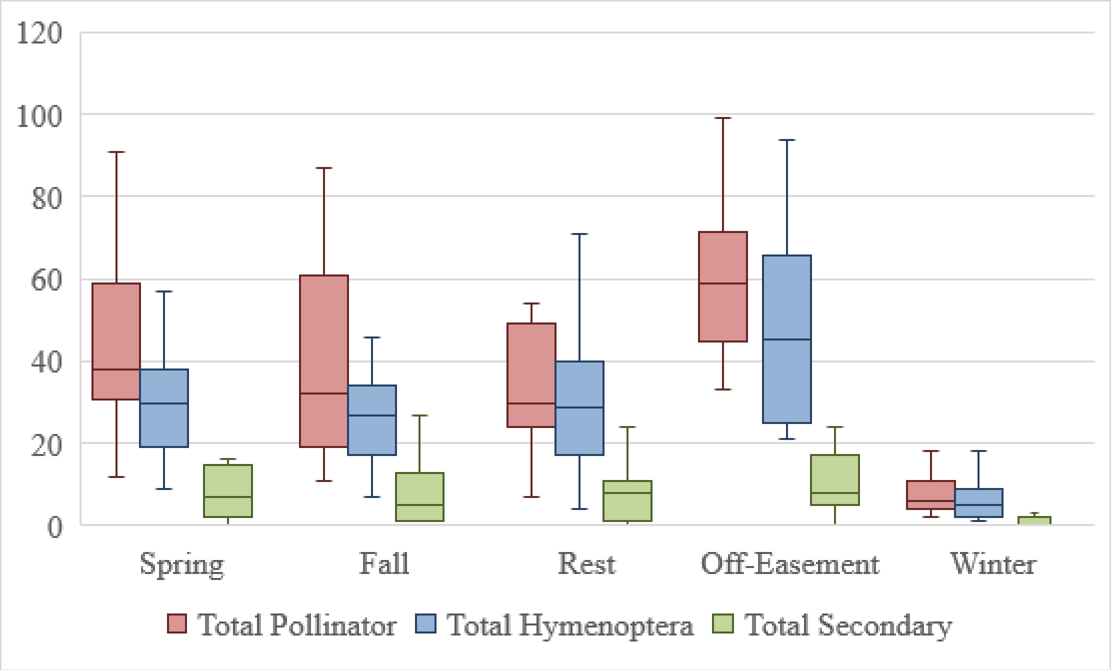

Figure 4. Boxplots of weekly total, Hymenoptera, and secondary pollinator activity-density collected from the 2018 field season east of Sidney, MT. The top and bottom whiskers indicate the largest and smallest observations within 1.5 interquartile ranges, the box represents the first and third quartile ranges, and the median is indicated by the horizontal line.

Table 3. Least squared means of weekly activity-density for all pollinators[1], Hymenoptera pollinators, and secondary pollinators[2] activity-density collected from colored (blue, white, yellow) traps during the 2016, 2017, and 2018 field seasons east of Sidney, MT. Letter groupings beside least squared means signify differences between the mean.

	2016			2017			2018		
	Total Pollinator	Hymenoptera	Secondary	Total Pollinator	Hymenoptera	Secondary	Total Pollinator	Hymenoptera	Secondary
Fall	44.86	42.43	-5.86 c	161.81	134.43	18.38	31.07 b	22.6 b	-0.40 a
Rest	46.87	42.84	-4.57 c	174.86	142.69	22.89	31.07 b	23.73 b	-1.67 ab
Spring	58.74	48.48	1.25 b	148.37	116.80	23.00	34.80 b	23.17 b	2.47 a
Off-Easement	60.98	60.53	-7.23 c	-	-	-	49.83 a	39.50 a	1.33 a
Winter	51.72	35.22	7.50 a	-	-	-	-0.16 c	-1.29 c	-6.98 b
SE	12.44	12.50	2.62	17.10	16.29	4.98	8.84	7.55	2.59
F-value	0.37	0.62	11.38	0.60	0.67	0.31	4.86	4.62	1.51
df	4, 64	4, 64	4, 64	2, 49	2, 49	2, 49	4, 49	4, 49	4, 49
P-value	0.83	0.65	< 0.01	0.56	0.52	0.73	< 0.01	< 0.01	0.05

[1] Pollinators is the summed total activity-density for all catches.

[2] Secondary pollinators is the summed total activity-density of all Lepidopteran families and Syrphidae (Diptera).

Averaged weekly over two-years (2016, 2018) of study, total pollinator and total Hymenoptera activity-density did not differ among five treatments. In 2017, only three of the five treatments were sampled and therefore were excluded from total treatment catches. Much variability in secondary pollinator activity-density was recorded among treatments with the greatest activity-density recorded in winter pastures when compared to fall ($t = 3.04$; df = 4, 256; $p < 0.01$), rest ($t = 2.88$; df = 4, 256; $p < 0.01$), and off-easement ($t = 2.61$; df = 4, 256; $p = 0.01$) pastures; however, activity-density between winter and spring and among spring and fall, rest and off-easement pastures did not differ (Table 4, Fig. 5).

Table 4. Averaged least squared means from two-years 2016 and 2018 of study of weekly total, Hymenoptera, and secondary pollinator activity-density collected from colored (blue, white, yellow) traps east of Sidney, MT.

	Total Pollinator	Hymenoptera	Secondary
Fall	38.89	33.59	-3.58 b
Rest	39.96	34.96	-3.29 b
Spring	42.48	31.26	2.04 ab
Off-Easement	50.18	44.09	-3.12 b
Winter	37.41	25.17	3.44 a
SE	10.22	9.11	2.56
F-value	0.54	1.16	3.89
df	4, 25	4, 25	4, 25
P-value	0.71	0.33	< 0.01

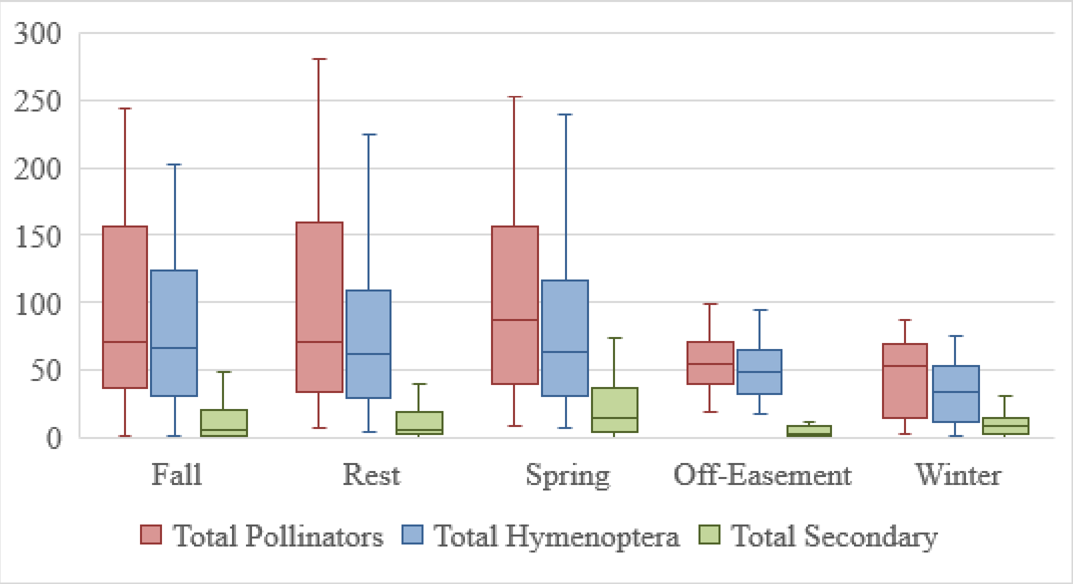

Figure 5. Boxplots of averaged weekly total, Hymenoptera, and secondary pollinator activity-density collected from the 2016, 2017, and 2018 field seasons east of Sidney, MT. The top and bottom whiskers indicate the largest and smallest observations within 1.5 interquartile ranges, the box represents the first and third quartile ranges, and the median is indicated by the horizontal line.

Genera

In 2016 at the Sidney site, 27 *Genera* were recorded and a total of 4,538 pollinator specimens. *Lasioglossum (Dialictus)* was collected the most in the fall pastures with 613 specimens when compared to rest, spring, and off-easement pastures and comprise 53% of the total Hymenopterans collected across all treatments. *Agapostemon* and *Halictus* also were abundant across all treatments making up 17.5% and 10% of total Hymenopteran catches across all treatments. Fall pastures were most productive with 1,065 Hymenopterans collected and spring was the least with 445 Hymenopterans. The *Genera Nomada*, *Dianthidium*, and *Panurginus* were only collected within spring pastures, whereas *Epeolus* was only collected in fall pastures and *Anthidium* in rested. Secondary pollinators were approximately 24% of total pollinators collected. Lepidopterans were more abundant than Syrphidae and both were most abundant in winter pastures (Table 5).

In 2017 at the Sidney site, 26 *Genera* were recorded with a total of 9,924 pollinator specimens. *Lasioglossum (Dialictus)* was most abundant in the rested pastures with 1,446 specimens when compared to fall and spring pastures and comprise 43% of total Hymenopterans collected. *Agapostemon* and *Halictus* were also abundant comprising 32% and 8% of total Hymenopterans collected. However, *Lasioglossum (s.str)* and *Eucera* are more abundant in 2017 when compared to 2016. Fall pastures were most abundant with 3,012 Hymenopterans collected and spring was the least with 2,268 Hymenopterans. *Anthidium*, *Perdita*, and *Hylaeus* were only collected within spring pastures, whereas *Hoplitis* was only in rested pastures. Secondary pollinators consisted of 18% of total pollinators collected. Lepidopterans were most abundant in 2017 than any other year and comprised of 96% of secondary pollinators collected. Both Lepidopterans and Syrphidae were most abundant in rested pastures (Table 5).

In 2018 at the Sidney site, 15 *Genera* were recorded with a total of 2,616 pollinator specimens. *Agapostemon* was the most collected *Genera* in 2018 comprising 39% of total Hymenopterans collected. *Agapostemon* was collected the most in off-easement pastures with 229 specimens. *Lasioglossum (Dialictus)* was the second highest collected specimen in which comprises 33% of total Hymenopterans and was highest also in off-easement pastures. The most Hymenopterans were collected within off-easement pastures with 582 specimens and the least in winter pastures with 47 specimens. The *Genera Sphecodes* was only recorded in rested pastures. Secondary pollinators consisted of 20% of total pollinators collected. Lepidopterans were highest in the spring pastures, whereas Syrphidae were highest in the off-easement (Table 5).

Table 5. Total primary pollinator Hymenoptera *Genus* (*subgenus*), and secondary pollinator Diptera family Syrphidae and order Lepidoptera from sampling seasons 2016, 2017, and 2018 at the Sidney, MT field site.

	2016[1]					2017			2018				
Genus (*subgenus*)	Fall	Rest	Spring	Off	Winter	Fall	Rest	Spring	Fall	Rest	Spring	Off	Winter
Lasioglossum (Dialictus)	613	569	234	418	-	1161	1446	869	144	142	158	238	14
Agapostemon	207	167	91	138	-	1048	822	737	220	169	176	229	18
Halictus	115	109	72	64	-	294	164	208	20	21	13	18	4
Melissodes	34	24	8	45	-	0	10	2	0	0	0	0	0
Lasioglossum (s.str)	30	30	9	26	-	185	130	95	32	57	23	18	3
Bombus	15	33	7	6	-	58	60	74	16	41	44	31	5
Diadasia	12	13	6	6	-	7	10	4	0	0	0	0	0
Andrena	9	9	2	9	-	95	87	78	22	27	41	9	1
Colletes	7	5	1	2	-	3	3	1	0	0	0	0	0
Lasioglossum (Shecodogastra)	6	3	2	5	-	13	18	11	4	7	5	4	0
Anthophora	4	3	0	1	-	14	8	5	0	2	0	5	0
Eucera	4	4	4	1	-	85	96	140	11	11	21	27	2
Megachile	3	2	0	0	-	4	2	1	1	0	0	1	0
Osmia	2	2	2	0	-	4	11	10	0	3	0	1	0
Lasioglossum (Evylaeus)	2	2	0	0	-	11	2	5	0	7	1	0	0
Epeolus	1	0	0	0	-	1	0	1	0	0	0	0	0
Protandrena	1	0	3	3	-	6	8	8	0	0	0	0	0
Nomada	0	0	1	0	-	10	8	8	1	2	0	0	0
Anthidium	0	1	0	0	-	0	0	1	0	0	0	0	0
Dianthidium	0	0	1	0	-	0	0	0	0	0	0	0	0
Panurginus	0	0	1	0	-	9	1	1	0	0	0	0	0
Dufourea	0	0	1	1	-	1	2	2	0	0	0	0	0

Table 5. Continued

	2016					2017			2018				
Genus (*subgenus*)	Fall	Rest	Spring	Off	Winter	Fall	Rest	Spring	Fall	Rest	Spring	Off	Winter
Sphecodes	0	1	1	1	-	2	2	1	0	1	0	0	0
Apis	0	0	0	0	-	1	1	4	1	1	3	1	0
Hoplitis	0	0	0	0	-	0	3	0	0	0	0	0	0
Perdita	0	0	0	0	-	0	0	1	0	0	0	0	0
Hylaeus	0	0	0	0	-	0	0	1	0	0	0	0	0
Family													
Diptera													
Syrphidae	26	32	38	7	137	18	34	21	11	12	10	14	1
Order													
Lepidoptera	25	43	44	16	160	557	571	549	116	98	156	110	6

[1] 2016 Pollinators collected from Winter pastures were only keyed to Family and thus not included in this table.

Vegetation Cover

In 2016 at the Sidney site, trapping location grass, flowering forb, and litter covers did not differ among treatments (Table 6). Forb cover was greatest in spring when compared to fall ($t = 4.40$; df = 4, 10; $p < 0.01$), rest ($t = 2.75$; df = 4, 10; $p = 0.02$), off-easement ($t = 2.41$; df = 4, 10; $p = 0.04$), and winter ($t = 4.40$; df = 4, 10; $p < 0.01$) pastures, which did not differ from one another. Trapping location bare ground was greatest in winter pastures when compared to spring ($t = 4.03$; df = 4, 10; $p < 0.01$), rest ($t = 3.73$; df = 4, 10; $p < 0.01$), and fall ($t = 2.57$; df = 4, 20; $p = 0.03$) pastures. However, bare ground in winter pastures did not differ from off-easement which did not differ from rest, fall pastures and spring pastures ($t = 2.28$; df = 4, 10; $p = 0.05$). Shrub cover was greatest in spring when compared to off-easement ($t = 2.40$; df = 4, 10; $p = 0.04$) pastures. Shrub cover in spring pastures did not differ from fall, rest, and winter which did not differ from off-easement (Table 6).

In 2017 at the Sidney site, trapping location shrub, flowering forb, bare ground, and litter did not differ. Grass cover was greatest in rest when compared to fall ($t = 2.52$; df = 2, 6; $p = 0.05$) pastures. Grass cover in spring did not differ from fall or rest pastures. Trapping location forb cover was greatest in fall pastures when compared to rest ($t = 3.41$; df = 2, 6; $p = 0.01$) and spring ($t = 2.55$; df = 2, 6; $p = 0.04$) pastures (Table 6).

In 2018 at the Sidney site, trapping location shrub, forb, flowering forb, and litter did not differ. Grass cover was greatest in winter pastures when compared to spring ($t = 3.09$; df = 4, 10; $p = 0.01$), off-easement ($t = 4.83$; df = 4, 10; $p < 0.01$), rest ($t = 4.59$; df = 4, 10; $p < 0.01$), and fall ($t = 3.40$; df = 4, 10; $p < 0.01$) pastures. Trapping location bare ground was greatest in off-easement when compared to spring (t = 2.45; df = 4, 10; p = 0.03) and winter (t = 3.51; df = 4, 10 ; p < 0.01) pastures; however, off-easement locations did not differ from fall or rest sampling locations which did not differ from spring and winter (Table 6).

Table 6. Weekly least squared mean comparisons of vegetation metrics for sampling seasons 2016, 2017, and 2018 at the Sidney, MT field site. Letter groupings beside least squared means signify differences between the mean.

	Grass	Shrub	Forb	Flowering Forb	Bare Ground	Litter
2016[1]						
Fall	28.83	5.00 ab	4.17 b	2.50	7.50 b	28.00
Rest	28.17	6.67 ab	8.17 b	2.50	4.17 b	33.50
Spring	31.17	10.50 a	14.83 a	3.33	3.33 b	21.17
Off-Easement	32.5	3.33 b	9.00 b	3.33	9.83 ab	24.00
Winter	28.83	8.17 ab	4.17 b	2.50	14.83 a	34.17
SE	5.24	2.10	1.71	0.43	2.02	7.26
F-value	0.12	1.73	6.56	1.12	5.31	0.62
df	4, 10	4, 10	4, 10	4, 10	4, 10	4, 10
P-value	0.97	0.22	< 0.01	0.40	0.01	0.66
2017						
Fall	16.88 b	5.21	9.17 a	2.50	7.00	46.14
Rest	23.48 a	6.88	4.62 b	2.74	12.02	36.79
Spring	19.71 ab	6.67	5.76 b	2.50	9.21	43.62
SE	1.85	1.19	0.94	0.09	1.83	3.58
F-value	3.20	0.58	6.30	2.21	1.88	1.83
df	2, 6	2, 6	2, 6	2, 6	2, 6	2, 6
P-value	0.11	0.59	0.03	0.19	0.23	0.24
2018						
Fall	14.73 b	3.21	8.00	3.00	6.28 ab	14.07
Rest	8.50 b	2.67	9.18	3.00	6.48 ab	11.48
Spring	16.30 b	3.17	6.00	2.50	4.83 b	17.77
Off-Easement	7.30 b	2.50	5.63	2.67	8.63 a	12.90
Winter	32.37 a	2.50	9.75	2.50	2.92 b	8.25
SE	3.59	0.20	1.05	0.32	1.10	2.94
F-value	7.33	2.86	2.88	0.59	3.43	1.27
df	4, 10	4, 10	4, 10	4, 10	4, 10	4, 10
P-value	< 0.01	0.08	0.08	0.67	0.05	0.34

[1] 2016 df represent one sampling date for field season.

Average over two years at the Sidney study site, coverage of forb and flowering forb did not differ among treatments. Trapping location grass cover was greatest in winter when compared to fall (t = 4.50; df = 4, 10; $p < 0.01$), rest (t = 4.16; df = 4, 10; $p < 0.01$), spring (t = 3.80; df = 4, 10; $p < 0.01$), and off-easement (t = 5.37; df = 4, 10; $p < 0.01$) pastures. Grass cover was reduced in off-easement when compared to spring (t = 2.54; df = 4, 10; $p = 0.03$) pastures; however, spring did not differ from rest and fall pastures, which did not differ from off-easement. Shrub cover was greatest in spring (t = 2.64; df = 4, 10; $p = 0.02$), and rest (t = 2.31; df = 4, 10; $p = 0.04$), when compared to off-easement; however, spring and rest did not differ from fall and winter which did not differ from off-easement. Bare ground cover was greatest (t = 2.19; df = 4, 10; $p = 0.05$) in rest when compared to winter pastures; however, rest did not differ from fall, spring, and off-easement, which did not differ from winter pastures. Plant litter at trapping locations did not differ among fall, rest, and spring pastures and between off-easement and winter pastures. Plant litter was greatest in fall compared to off-easement (t = 3.34; df = 4, 10; $p < 0.01$) and winter (t = 3.37; df = 4, 10; $p < 0.01$), in spring pastures compared to off-easement (t = 3.25; df = 4, 10; $p < 0.01$) and winter (t = 3.28; df = 4, 10; $p < 0.01$), and in rest compared to off-easement (t = 2.28; df = 4, 10; $p = 0.05$) and winter (t = 2.37; df = 4, 10; $p = 0.04$) (Table 7).

Table 7. Averaged weekly least squared means across all three sampling years 2016, 2017 and 2018. Letter groupings beside least squared means signify differences between means.

	Grass	Shrub	Forb	Flowering Forb	Bare Ground	Litter
Fall	16.97 bc	4.43 ab	8.33	2.69	6.76 ab	32.41 a
Rest	18.08 bc	5.24 a	6.65	2.82	9.29 a	26.80 a
Spring	19.28 b	5.62 a	6.55	2.56	7.08 ab	31.95 a
Off-Easement	11.50 c	2.64 b	6.19	2.78	8.83 ab	14.75 b
Winter	31.67 a	3.63 ab	8.63	2.50	5.30 b	13.43 b
SE	2.77	1.01	1.17	0.24	1.47	4.79
F-value	7.80	2.23	1.55	0.54	1.64	5.48
df	4, 10	4, 10	4, 10	4, 10	4, 10	4, 10
P-value	< 0.01	0.14	0.26	0.71	0.24	0.01

Discussion

Sidney, Montana is a mostly an agricultural crop and livestock dependent rural community that lays within the great plains mixed grass prairie biome. This region received an average annual rainfall of 363.73 mm from 1981-2010 (NOAA, Accessed 25 April 2019). At the Sidney site, vegetation metrics of forb and flowering forb did not differ when data were combined over the three years (2016, 2017, and 2018) of study. Grass cover in my sampling locations of fall, rest, and spring pastures was moderate. However, I am also seeing the highest levels of litter within these pastures which appears to be function of a rest-rotation grazing program (Jacobo *et al.* 2006; Vecchio *et al.* 2019). While this program allows for an accumulation of litter, I speculate lower litter accumulation in off-easement pastures may be a function of its non-enrollment in the program, result of higher stocking rates or a more frequent grazing rotation; all serving as a function of no rest period as cattle feed at high intensities within pastures. This may also be true with winter pastures. When analyzed on a yearly basis, 2016 vegetation metrics indicate that spring pastures were highest in non-flowering forb resources and winter and off-easement pastures were highest in bare ground. Year 2017 displays fall pastures are highest in non-flowering forb resources. In 2018, bare ground was highest in fall, rest, and off-easement pastures. Interestingly, these three metrics (i.e., forb and flowering forb resources and bare ground), in addition to litter, are often the most influential to pollinator survival as they provide either food or nesting resources. A publication from the USDA National Agroforestry Center indicates that, while native bee nesting specifics are not well-known, there are some elemental characteristics that are thought to support many species to some capacity (National Agroforestry Center, 2007). These include the incorporation of nearby forb and shrub pollen resources, open bare ground areas for ground nesting, and the exclusion of thatch which acts as a deterrent against native bees accessing the soil surface to construct nesting galleries (National Agroforestry Center, 2007). These habitat characteristics may explain some of the variability I am seeing in the pollinators present. For example, Lepidopterans engage in puddling behaviors to retrieve necessary mineral resources (Cue, 2017) and in 2016, were highest in winter pastures that was also highest in bare ground. *Lasioglossum (Dialictus)* nesting behaviors are often recorded in bare ground sites (Kimoto *et al.*, 2012; Wilson & Carril, 2016) and was numerically recorded most often in the rested pastures in which were highest in bare ground in 2017.

When analyzing pollinator activity-densities, sampling years 2016 and 2018 included the off-easement and winter pastures for sampling. Therefore, when looking at multi-year trends, 2017 was not included. When pollinator activity-density was combined for sampling years 2016 and 2018, there were no significant differences in the total pollinator or total Hymenopteran categories among treatments while secondary pollinator activity-density was highest in winter pastures. During 2016 and 2018, differences occurred in winter and off-easement secondary pollinator activity-densities. In year 2016, secondary pollinator activity-density was highest in winter pastures which differed from 2018 when activity-density was highest in off-easement pastures. In 2016, sampling started almost exactly one month later than and included two more weeks of sampling winter pastures and one more week of off-easement pastures than what occurred in 2018. In addition, 2016 was a more complete year of sampling with six weeks of pollinator trapping in winter pastures while 2018 was more temporally limited with three weeks of sampling collections. This is evident when comparing 2016 and 2018 winter pastures in that the 2016 secondary pollinator activity-density in winter pastures was 6.5% of total pollinator catches and only 0.27% of total pollinator catches in 2018 suggesting that winter pasture collections were largely influenced by the reduced sampling effort.

Most pollinators become active and begin to emerge from their overwintering sites when daily temperatures exceed 15˚C; however, some species of pollinator are active earlier in spring when daily temperatures reach 10˚C (Stubbs & Coverstone, 2004). In addition to the sampling time period being shorter in 2018, the sampling also occurred earlier when the total pollinator activity-density was lower resulting in fewer trap encounters and captures. In addition, the winter pasture cattle turn-in and take-out dates over 2016 and 2018 stay relatively constant: cattle move into winter pastures either early October or November the year before and rotate out into spring pastures early June

(Table 1). Vegetation data combined with cattle rotation dates provides insight to 2016 secondary pollinators captures. Winter pasture sampling during 2016 began 20-June when cattle had just been removed from the winter pastures. In contrast, 2018 winter pasture sampling began the week of 22-May, close to one month earlier. Sampling began in 2016 when degree days had accumulated to 1194.1 while 2018 sample began at 414.9-degree days. Given this, along with winter pastures experiencing a longer period of cattle grazing prior to trapping which would have decreased thatch and increase bare ground, it is likely that secondary pollinators were more active and returning to winter pastures for reproductive purposes. One study by Balmer and Erhardt (2001) suggested that some butterfly species prefer early seral and/or recently livestock grazed landscapes while Griebeler and Seitz (2002) report that some butterfly species may rely on high grazing pressure in order to maintain their host plant (*Thymus).* Other studies suggest that hover flies (Diptera: Syrphidae) are negatively associated with high stocking rates (Sjödin *et al*., 2008; Lazaro *et al*., 2016), removal of cattle may have allowed hover flies to begin to reestablish in grazed areas.

Annual precipitation may also be a contributing factor of 2018 pollinator activity-density. A 30-year average for the Sidney, MT area indicates the highest precipitation during this study was in 2018 with 339.59 mm and was the lowest in 2017 with 209.29 mm (uspest.org). The high precipitation in 2018 may have contributed to differences in forb cover. While forb cover as measured by my Daubenmire frames is not representative of the available nectar and pollen resources, it may be an indicator that increased forb ground cover in the rest and fall pastures may be an attractant to native pollinators as various forb species may be flowering throughout the pasture.

Lazaro *et al*. (2016) suggested that bee abundance and richness was highest at intermediate grazing mid-season and was positively associated with intensity of grazing late season. Lazaro *et al*. (2016) also note that early season bee abundance was negatively affected by grazing and suggests this may be due to a specific species of bee outcompeting others. Various other studies have documented increased bee abundance with grazing intensity (Carvell, 2002; Vulliamy *et al*., 2006; Morandin *et al*., 2007) as it is proposed that this may reflect heightened nesting resources (Vulliamy *et al*., 2006).

In my study, the activity-density of *Lasioglossum* (*Dialictus*) and *Agapostemon,* supports these findings. These two bee *Genera* are well-known for their abundance and ground nesting behaviors within North America (Wilson & Carril, 2016). I recorded a similar trend with high activity-density recorded from all treatments at the Sidney site over all three study years. As noted in Kimoto *et al*. these two sweat bee *Genera* have been recorded to benefit from livestock grazing (2012) and my results support this conclusion. Additional literature also suggests sweat bee (i.e., *Lasioglossum*, *Agapostemon*, *Halictus*) nesting behaviors indicate a preference toward compacted soils and bare ground for nesting sites (Potts & Willmer, 1997, 1998; Kimoto *et al*., 2012). Although I did not measure soil compaction in this study, one result of livestock grazing can be in more compact soils (Kimoto *et al*., 2012).

The *Genera Lasioglossum* contains about 1,800 species globally that are generally 3-8 mm long and range in sociality from solitary to primitively eusocial (Wilson & Carril, 2016). These bees are found from early spring through fall seasons (Wilson & Carril, 2016). While this *Genera* generally have short flight distances corresponding with their size (Waddington, 1979), Kimoto *et al.* note that they did not appear to be sensitive to livestock grazing due to their limited flight to adjacent fields with potentially more resources (2012). While I continually recorded *Lasioglossum (Dialictus)* in all pastures, their early emergence and lack of sensitivity to livestock grazing may allow this *Genus* to out-compete other less abundant *Genera* and thrive in many environments.

The *Genera Agapostemon* is much larger in body size than *Lasioglossum* and capable of retrieving a larger suite of floral resources due to their ability for longer flight distances. In 1979, Waddington documented direct flight behaviors to dense patches of floral resources in order to maximize their flight energy costs. Like *Lasioglossum*, *Agapostemon* nests within the ground and has been documented to nest communally (Wilson & Carril, 2016) and maintain ‘house-keeping’ behaviors of keeping nesting sites free of debris (Abrams & Eickwort, 1981; NRCS, 2009). While *Agapostemon* are not truly eusocial, they may share nest entrances to reduce the risk of their nests being parasitized by cuckoo bees. Although it has been documented that livestock grazing may reduce necessary floral resources, and directly trample native bee nesting sites, reduce soil stability, and compress soils (Gess & Gess, 1983; Sugden, 1985; Potts *et al*., 2005; Vulliamy *et al*., 2006; Williams *et al*., 2010; Kimoto *et al*., 2012), the sweat bees and many other native bees generally benefit from these disturbances and nesting site changes (Michener, 2007; Winfree *et al*., 2009; Wilson & Carril, 2016).

The *Genus Halictus* is another sweat bee that was well documented by trap captures in all years and in each treatment during this study. All known species of *Halictus* are generalist feeders and produce many generations per year (Wilson & Carril, 2016). *Halictus* is unique in that members of this *Genus* are solitary nesting bees in cooler temperatures when floral resources are limited, and nest eusocially in warmer temperatures where (uncommonly) all females in a nest lay eggs and then fall subordinate to a queen (Wilson & Carril, 2016). Bombus is another eusocial *Genera* commonly found at the Sidney site and is also a generalist feeder that is active most of the year. These bees are known for their method of buzz pollination and production of larger fruit crops, faster fruit set, and larger yields (Wilson & Carril, 2016). The nesting behaviors and activity of sweat and bumble bees may have contributed to the activity-densities recorded from treatment sampling. My data suggests that *Halictus* is more associated with fall grazed pastures while *Bombus* is more associated with pastures either rested from or low intensity livestock, or yak as documented in Xie *et al.*, grazing (Xie *et al.*, 2008; Kimoto *et al.*, 2012). This may provide insights into habitat preferences in relation to cattle disturbance.

Lastly, the *Genus Eucera* was also a common catch in all treatments over the three-year course of this study that is a commonly found bee that is active in early spring through July (Wilson & Carril, 2016). Most species of this *Genus* are generalist feeders (Wilson & Carril, 2016). Like those above, *Eucera* are ground nesting bees that builds small mounds around nests within sandy or clayey soils (Wilson & Carril, 2016).

Rest-rotation grazing can be an effective land management tool and was put into place at my Sidney, MT field sites in order to enhance grouse and other bird populations. When combining information on environmental factors and biological characteristics of the most common native bees, we can understand, to some extent, the influence of this management technique on pollinator conservation. While this management system can benefit both the flora and fauna of an area (Gibson *et al*., 1992; Sayre *et al*., 2012; Enri *et* *al*., 2017; Lyons *et al*., 2017), it appears that it is suited for wetter biomes where precipitation allows consistent regrowth of plant matter and maintains animal intake (Bailey & Brown, 2011). While this study did not specifically test for this, the high precipitation and lack of many differences between the key vegetation in Sidney might suggest rest-rotation grazing management is superfluous in an area that can maintain quality vegetation during growing seasons.

CHAPTER THREE

EFFECTS OF LIVESTOCK GRAZING ON POLLINATORS- ROUNDUP

Introduction

Endemic to western North America, sagebrush ecosystems provide many pertinent ecological, hydrological, and recreational functions (Perfors *et al*., 2003; Connelly *et al*., 2004; Davies *et al*., 2007). While their original range has contracted by approximately 50% due primarily to conversion to farmland and climatic factors (Connelly *et al*., 2004; Schroeder *et al*., 2004; Hagen *et al*., 2007), sagebrush ecosystems continue to provide resources for many obligate species (Homer *et al*., 2015). One of these species, the GSG, relies on these shrublands for cover, nesting, and food resources. Greater sage-grouse chicks rely on herbaceous forbs and arthropods as a food source of which their diet comprises over 60% arthropods during the first 21-28 days of a chick's life (Klebenow & Gray, 1968; Peterson, 1970). While many beneficial and pest insects inhabit these vast ecosystems, native pollinators (i.e., Lepidoptera larvae) are not only an important aspect to grouse chick diets, but a predominant organism in sage ecosystems that are positively associated with herbaceous forbs, a staple of both adult and chick sage grouse (Kasten *et al.*, 2016).

One study by Cook *et al*. (2011) found 12 species of bumblebees in a single Idaho sagebrush site signifying the importance of these habitats to native pollinator diversity and abundance. Additionally, sagebrush landscapes are also fertile and rich grazing lands necessary for economic stability of many rural communities in the western U.S. While the relationships between plants, animals and grazing livestock may oscillate depending on conditions, the richness and abundance of native bees critical to this ecosystem is also consequently impacted and therefore an equilibrium must exist among all components for native bees to survive. As result of this, it has led to the initiating of this project to monitor native bees' populations in one central Montana sagebrush steppe habitat. From this project, we can determine if pasture enrollment in the SGI has a positive impact on pollinator activity-density in contrast to pastures that are grazed according to land lessee, or to pastures that have not experienced livestock grazing in at least seven years. With this guiding research question, I hypothesize that enrollment in the SGI will have a positive impact on activity-densities.

Study Area

Roundup, MT is a sagebrush dominated ecosystem that varies in topography, soils, habitat type, and biodiversity (Connelly *et al.*, 2000). Roundup, MT is located approximately 80 km north of Billings, MT and receives an average of about 359 mm of precipitation each year with an average high temperature of 17.3˚C and a low of 1˚C (USclimatedata.com). This study took place in Musselshell County, an area that is largely privately owned, managed by BLM, or managed by the state of Montana (Woodward *et al.*, 2011). While this area is primarily used for livestock grazing, in 1941 three separate allotments were designated as the Lake Mason Wildlife Refuge (hereafter LMWR) and consists of mixed-grass prairie and sagebrush ecosystems (USFWS, 2014). The northern unit of LMWR contains pastures that are our 'non-grazed' sampling sites. Neighboring the LMWR are pastures that are either enrolled within an NRCS established

‘Sage Grouse Core Area’ pastures or pastures that are not. Pastures established within the Sage Grouse Core Area are enrolled within the SGI, a management area that focuses on wildlife conservation through sustainable ranching (USDA, 2016). These pastures are designated as ‘enrolled’ pastures for sampling within this study. The enrolled pastures undergo an 18-month rest rotation grazing management; however, are not grazed and within the rested state when sampled. Lastly, pastures that are neither within the LMWR or within the SGI program are designated as our ‘non-enrolled’ pastures. These pastures are either privately or publicly owned and prescribed management is as designated by the lessee.

Methods

Pollinator Surveys

Assessment of bee diversity and abundance is often measured with malaise traps, vane traps (McCravy & Ruholl, 2017), sweep-netting, and pan traps (Popic *et al.*, 2013) in a variety of ecosystems. However, the pan trap method is most commonly used for pollinator studies as it is cost effective and a time efficient passive sampling method (Leong & Thorp, 2001; Westphal *et al.*, 2008) with low collector bias (Saunders & Luck, 2013). Common items used as pan traps are plastic deli bowls, pans, or cups that are filled with a mixture of soapy water (Droege, 2012) or a 50:50 mixture of propylene glycol and water (Packer, 2012) to act as a killing agent and/or preservation agent. Both mixtures decrease surface tension, generally drown a landing insect within a minute, and have the potential to collect insect by-catch (Droege, 2015). One study recorded that 33% of bees escape pan traps regardless of the drowning solution (Droege, 2015). Pan traps which are colored inside and out may be used to mimic the visual appearance of floral resources to enhance bee visitation rates (Saunders & Luck, 2013). Blue, white, and yellow pan traps are most commonly used (Leong & Thorp, 2001) and align well with floral resources of our Roundup location.

Weather and Degree Days

Wind, temperature, and precipitation averages were retrieved from the Horsethief, MT HORM8 RAWS weather station located at 46.4256 and -108.6742 (USpest.org). While 30-year precipitation averages were not available for the Horsethief, MT HORM8 RAWS weather station, averages were found on the NOAA Climate Data online web page at nearby weather station Roundup, MT 15 SW GHCND: USC00247220 (US Climate Data, 2019). Degree days were calculated by selecting the Horsethief, MT HORM8 RAWS weather station and setting lower threshold to 0˚C and the upper to 30˚C utilizing the single sine method (USpest.org).

Setting Traps

Our methods followed Droege (2012) who outlined bee bowl sampling. Nine pastures were sampled yearly at the Roundup location. In 2016, 2017, and 2018, and in the approximate center of each pasture (i.e., experimental unit), a single transect of pollinator traps was randomly deployed within each pasture. Each transect contained a total of nine plastic cups (16 oz. Solo®, Lake Forest, IL): three blue, three yellow, and three white extending in a straight line, approximately three meters apart (Droege *et al*., 2010). For sampling years 2016 and 2017, each trap was set on the ground in shallow hole leaving 90% of the colored trap exposed. Trap holes were generated with a standard one-piece soil auger (AMS®, American Falls, ID), traps were placed approximately 2.5 cm deep in the hole and then backfilled with remaining soil to provide stability for the cup. In order to keep traps highly visible, vegetation was flattened down and away from traps during the growing season. Traps were replenished weekly from a one-gallon plastic jug filled with tap water and one tablespoon of unscented dish soap (Dawn®, P&G, Cincinnati, OH), where each trap was filled approximately ¾ full. Additional eight-gallon water containers and dish soap were stored in the truck in order to refill jugs for each transect.

Collecting

As denoted above, traps were deployed seven days per week and the contents were always collected and traps reset on the same day. Replicate catches from the three similar colored traps were combined in Whirl-Paks® (Nasco, Fort Atkinson, WI) pre-labeled with cup color, date, and pasture locality. Large non-pollinator debris, such as rocks, sticks, leaves, etc. were removed from each sample on location. Whirl-Paks® were then stored as is in a cooler. Traps were visually inspected for cracks or damage and re-filled ¾ full of soapy water and replaced in the same color pattern and transect as initially determined.

Processing Samples

Whirl-Paks® were returned to the Montana State University laboratory where samples were cleaned. Guidelines for cleaning and processing samples are detailed in *The Handy Bee Manual* (Droege, 2015) and modified as necessary. My process was transferring Whirl-Pak® contents to a fine mesh soil strainer, removing additional large debris, rinsing with cool water, and finally replacing in separately marked Whirl-Paks® with enough 95% ethanol solution to cover and preserve the specimens. At a later date, preserved specimens were transferred to a petri dish where members of Lepidoptera and Syrphidae were identified and counted. Hymenopteran specimens containing branched hairs were pulled from samples, labeled, and kept in a petri dish in a freezer (-17 °C) until further processing could occur. Branched haired bees were placed in a 473 mL Ball® mason jar (Ball Corporation, Broomfield, CO) with one drop of unscented dish soap (Dawn®, P&G, Cincinnati, OH) and approximately 29 mL of water. The mason jar top was secured, and the contents shaken for about 60 seconds to remove additional small debris, clean matted hairs, and generally prepare the specimens for identification. Cleaned specimens were transferred to a paper towel and excess water blotted away. After about 3-minutes, bees were transferred to a dry mason jar containing three small balled-up pieces of paper towel (to assist in soaking up excess water and aid in fluffing the fine hairs on the bee bodies) and the jar opening was secured with a rubber band and fine mesh net top. The contents were dried by shaking the jar while forcing warm air over the specimens using a commercial hair dryer (Conair®, Conair Corporation, Stamford, CT). After this drying process, specimens were pinned, labeled, and stored in airtight specimen drawers until identified. Specimens were identified to *Genus* utilizing *The Bee Genera of North and Central America* key (Michener *et al*., 1994) and individuals were recorded by color, date, and location within an excel spreadsheet.

Vegetation Cover

Vegetation cover and composition were measured weekly upon collection of pollinator traps as established by Daubenmire (1959). Working at the beginning of a pollinator trap transect, a 20 cm x 50 cm frame was placed at five intervals three meters apart extending in a single direction away from the transect. Using the quadrat markings on the frame, vegetation was denoted as either forb, flowering forb, shrub, grass, litter, lichen, prickly pear, or bare ground on a scale of 1-6 indicating an approximate percentage range of coverage, and finally the midpoint of range. Table 8 provides the range of coverage associated with each cover class (Daubenmire, 1959).

Table 8. Daubenmire cover class range of coverage percentages.

Cover Class	Range of Coverage	Midpoint of Range
1	0-5%	2.5%
2	5-25%	15.0%
3	25-50%	37.5%
4	50-75%	62.5%
5	75-95%	85.0%
6	95-100%	97.5%

Statistical Analysis

Pollinator count data and Daubenmire vegetative cover data were analyzed within sampling year and combined over study years. All data were fit to a generalized linear mixed model with sampling location as the random variable (SAS, 2002). Akaike information criterion (AIC) values were used to assess the relative quality of candidate model distribution where smaller AIC values indicate a better model fit and subsequent explanation of the data. This process resulted in pollinator count data being fit to a negative binomial distribution and Daubenmire vegetative cover data being fit to a normal distribution within the generalized linear mixed model. Differences among treatment least squared means were calculated using the least significant difference test.

Results

We collected a total of 13,692 pollinator specimens over the course of three collecting seasons from 2016 to 2018 at the Roundup, Montana field location. With a total of 17 sampling weeks spanning from May to July over a three year period, we collected a total of 27 Hymenoptera *Genera* from various grazed and non-grazed pastures that are either enrolled in the SGI program with a 18-month rest-rotation (SGI), pastures that are not enrolled in the SGI program and grazed according to either the landowner or leasee (NSGI), or pastures not grazed by livestock in at least seven years (LMWR).

Activity-Density

In 2016 at the Roundup site, least squared mean comparisons of treatment level weekly pollinator activity-density, a function of (pollinator activity across a landscape) x (population density) (Hokkanen & Holopainen 1986), was greatest ($t = 2.49$; df = 2, 6; $p = 0.02$) in non-enrolled when compared to non-grazed pastures, while activity-density in enrolled pastures did not differ from either non-enrolled or non-grazed pastures. Hymenoptera activity-density was greatest in enrolled ($t = 1.95$; df = 2, 6; $p = 0.05$) and non-enrolled (t = 2.61; df = 2, 6; p = 0.01) pastures when compared to non-grazed pastures. Secondary pollinator activity-density did not differ among treatments (Table 9, Fig. 7).

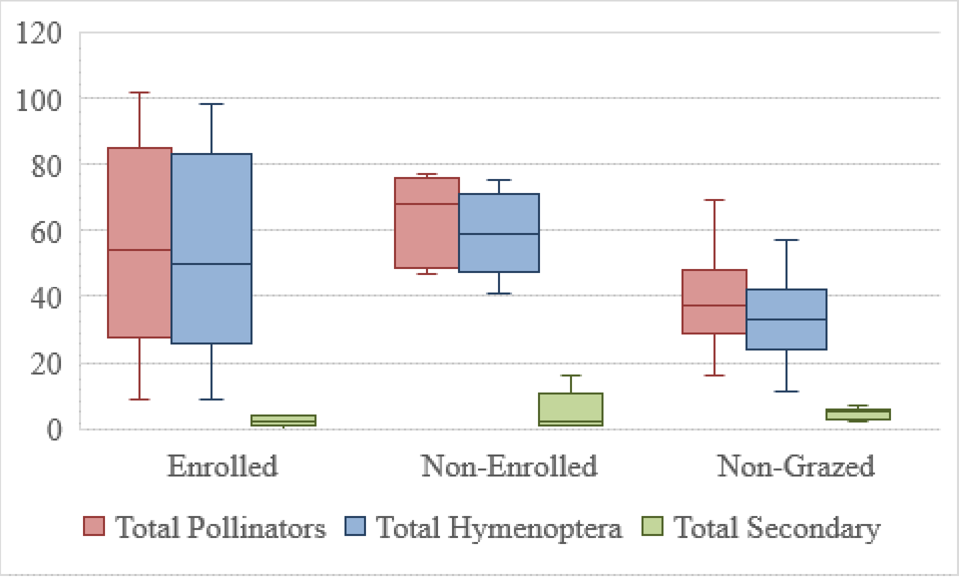

Figure 6. Boxplots of weekly total, Hymenoptera, and secondary pollinator activity-density collected from the 2016 field season north of Lavina, MT. The top and bottom whiskers indicate the largest and smallest observations within 1.5 interquartile ranges, the box represents the first and third quartile ranges, and the median is indicated by the horizontal line.

In 2017 at the Roundup site, least squared mean comparisons of treatment level total pollinator activity-density was greatest in enrolled (t = 4.13; df = 2, 6; $p < 0.01$) and non-enrolled (t = 2.43; df = 2, 6; p = 0.02) pastures when compared to non-grazed pastures; however, they did not differ from each other. Hymenoptera activity-density also was greatest in enrolled (t = 4.61; df = 2, 6; $p < 0.01$) and non-enrolled (t = 3.54; df = 2, 6; $p < 0.01$) pastures when compared to non-grazed pastures; but they did not differ from each other. Secondary pollinator activity-density was greater in enrolled (t = 2.93;

df = 2, 6; *p* < 0.01) and non-grazed (*t* = 3.97; df = 2, 6; *p* < 0.01) when compared to non-enrolled pastures but they did not differ from each other (Table 9, Fig. 8).

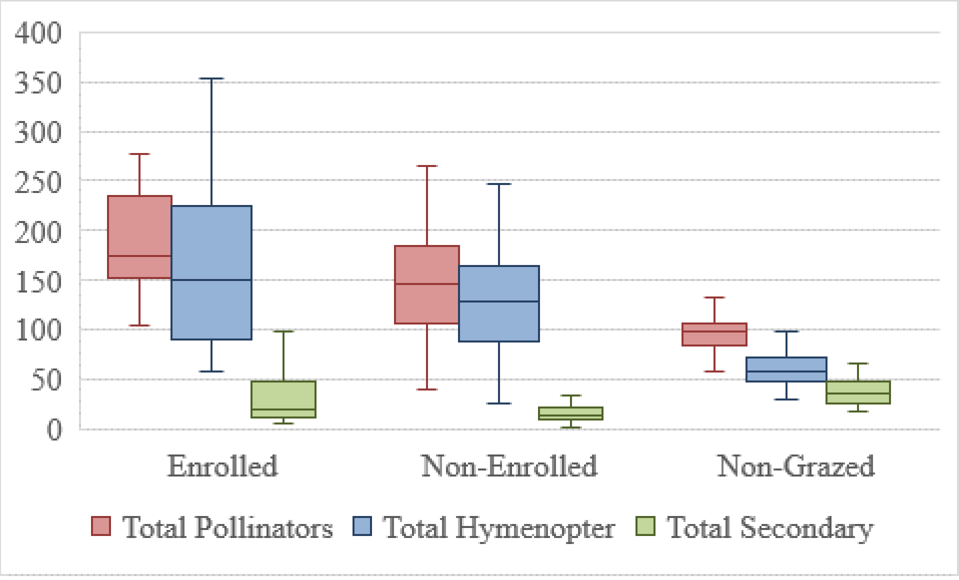

Figure 7. Boxplots of weekly total, Hymenoptera, and secondary pollinator activity-density collected from the 2017 field season north of Lavina, MT. The top and bottom whiskers indicate the largest and smallest observations within 1.5 interquartile ranges, the box represents the first and third quartile ranges, and the median is indicated by the horizontal line.

In 2018 at the Roundup site, least squared mean comparisons of treatment level activity-density did not differ in total pollinators, Hymenoptera, and secondary pollinators (Table 9, Fig 9).

Table 9. Least squared means of weekly total, Hymenoptera, and secondary pollinator activity-density collected from colored (blue, white, yellow) traps during the 2016, 2017, and 2018 field seasons north of Lavina, MT. Letter groupings beside least squared means signify differences between the mean.

	2016			2017			2018		
	Total Pollinator	Hymenoptera	Secondary	Total Pollinator	Hymenoptera	Secondary	Total Pollinator	Hymenoptera	Secondary
Enrolled	50.10 a	47.26 ab	-5.54	182.57 a	151.33 a	22.23 a	47.76	41.64	-1.76
Non- Enrolled	65.07 a	60.06 a	-3.89	143.43 a	127.86 a	6.71 b	47.64	39.78	0.12
Non- Grazed	30.55 b	25.37 b	-3.81	87.24 b	50.52 b	27.71 a	41.87	30.40	-1.13
SE	10.25	9.80	1.26	16.33	15.46	3.74	7.97	7.45	1.70
F- value	3.15	3.55	0.78	8.61	11.64	8.47	0.19	0.27	0.32
df	2, 24	2, 24	2, 24	2, 54	2, 54	2, 54	2, 34	2, 34	2, 34
P-value	0.02	0.04	0.45	< 0.01	< 0.01	< 0.01	0.83	0.77	0.73

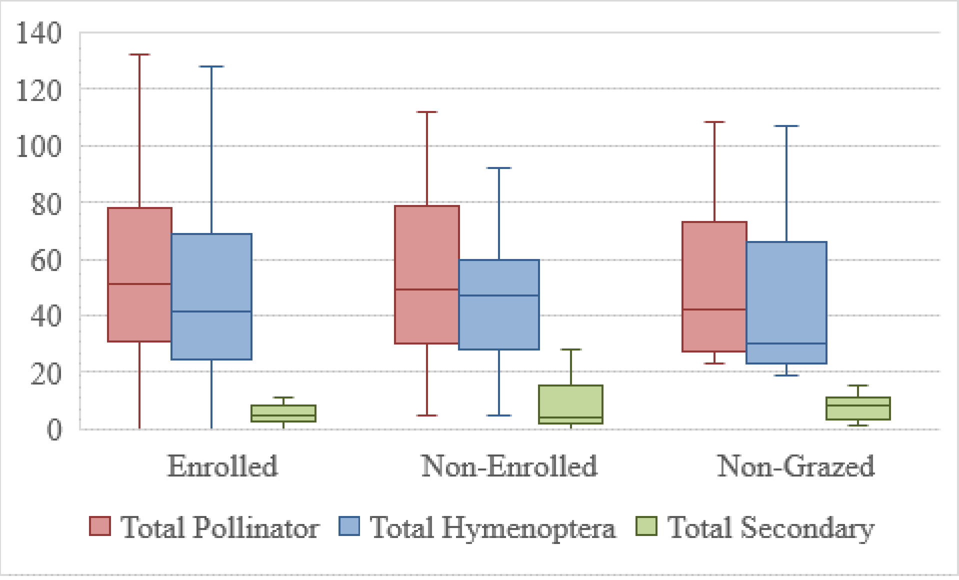

Figure 8. Boxplots of weekly total, Hymenoptera, and secondary pollinator activity-density collected from the 2018 field season north of Lavina, MT. The top and bottom whiskers indicate the largest and smallest observations within 1.5 interquartile ranges, the box represents the first and third quartile ranges, and the median is indicated by the horizontal line.

Averaged over the three years of study, total pollinator weekly activity-density was greatest in enrolled (t = 3.60; df = 2, 24; p < 0.01) and non-enrolled (t = 2.76; df = 2, 24; p < 0.01) when compared to non-grazed; but they did not differ from each other. Hymenoptera activity-density was also greatest in enrolled (t = 4.21; df = 2, 24; p < 0.01) and non-enrolled (t = 3.71; df = 2, 24; p < 0.01) when compared to non-grazed; but they did not differ from each other. Secondary pollinator activity-density was greatest (t = 2.44; df = 2, 24; p = 0.02) in non-grazed when compared to non-enrolled pastures while activity-density in enrolled pastures did not differ from non-grazed and non-enrolled pastures (Table 10, Fig. 10).

Table 10. Least squared means of combined 2016, 2017, and 2018 weekly total, Hymenoptera, and secondary pollinator activity-density collected from colored (blue, white, yellow) traps north of Lavina, MT. Letter groupings beside least squared means signify differences between the mean.

	Total Pollinators	Hymenoptera	Secondary
Enrolled	107.29 a	91.02 a	7.71 ab
Non- Enrolled	96.91 a	85.93 a	2.46 b
Non- Grazed	59.47 b	39.47 b	11.13 a
SE	9.74	9.00	2.55
F-value	7.13	10.61	3.01
df	2, 24	2, 24	2, 24
P-value	< 0.01	< 0.01	0.05

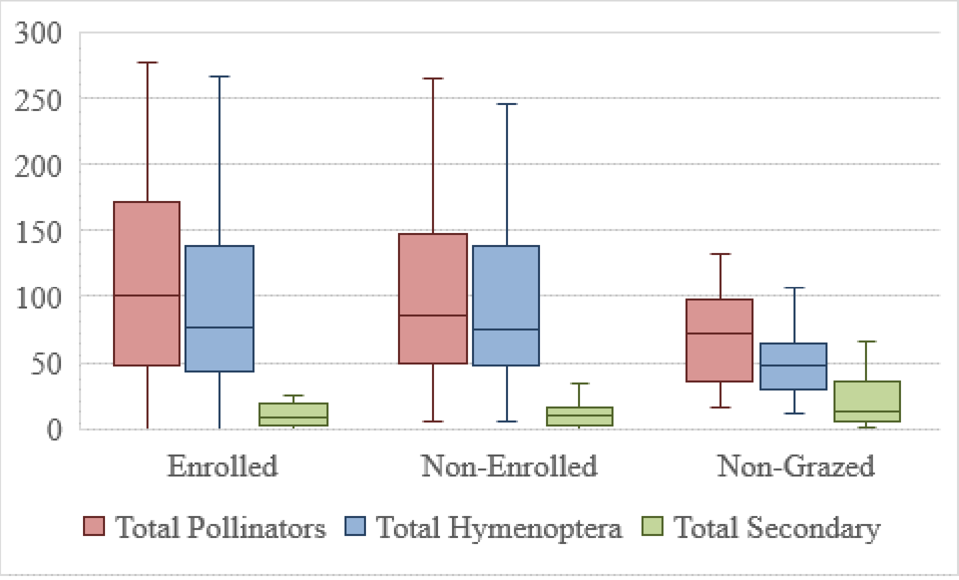

Figure 9. Boxplots of averaged weekly total, Hymenoptera, and secondary pollinator activity-density collected from the 2016, 2017, and 2018 field seasons north of Lavina, MT. The top and bottom whiskers indicate the largest and smallest observations within 1.5 interquartile ranges, the box represents the first and third quartile ranges, and the median is indicated by the horizontal line.

<u>*Genera*</u>

In 2016 at the Roundup site, 27 *Genera* were recorded and a total of 2,078 pollinator specimens. *Lasioglossum (Dialictus)* was collected the most in enrolled pastures with 269 specimens when compared to non-enrolled and non-grazed pastures and comprise 33% of total Hymenopterans collected across all treatments. *Agapostemon* and *Eucera* also were abundant across all treatments making up 28% and 11% of total Hymenopteran catches across all treatments. Enrolled pastures collected the most Hymenopterans with 719 specimens and non-enrolled was the least with 620. The *Genera Nomada*, *Ashmeadiella, Hoplitis, Hylaeus,* and *Stelis* were only collected within non-grazed pastures, whereas *Calliopsis* was only collected in non-enrolled pastures. The *Genus Macrotera* was only present in 2016 and not commonly found in Montana. Secondary pollinators were approximately 12% of total pollinators collected. Lepidopterans were more abundant than Syrphidae and both were most abundant in non-grazed pastures (Table 11).

In 2017 at the Roundup site, 24 *Genera* were recorded and a total of 9,290 pollinator specimens. *Agapostemon* was the most abundant across all three sites with a total of 2,137 specimens which comprises about 28% of total Hymenopterans collected in 2017. *Lasioglossum (Dialictus)* was the second most abundant with 2,050 specimens and accounted for approximately 27% of total Hymenopterans collected. *Andrena* was the most abundant in 2017 when compared to 2016 and 2018 with 588 specimens. Enrolled pastures collected the most Hymenopterans with 3,367 specimens and non-grazed pastures the least with 1,250 specimens. The *Genera Melissodes* and *Ceratina* were only present in enrolled pastures. Secondary pollinators were approximately 19% of total pollinators collected. Both Lepidoptera and Syrphidae were the most abundant in the enrolled and the least amount were caught in non-enrolled pastures. Catches for both secondary pollinators were highest in 2017 when compared to 2016 and 2018.

In 2018 at the Roundup site, 16 *Genera* were recorded and a total of 2,315 pollinator specimens. *Agapostemon* was the most abundant across all three sites with a total of 420 specimens which comprises about 20% of total Hymenopterans collected in 2018. *Lasioglossum (Dialictus)* and *Andrena* were also very abundant accounting for 19% and 16% of 2018 total Hymenopterans. Enrolled pastures collected the most Hymenopterans with 707 specimens and non-grazed the least with 636. *Melissodes* and *Hoplitis* were only collected within the non-enrolled pastures. Secondary pollinators were approximately 13% of total pollinator catches. Lepidoptera and Syrphidae were collected the most within non-grazed pastures, however Lepidoptera was the least in enrolled pastures and Syrphids were equally recorded for both enrolled and non-enrolled pastures.

Table 11. Total seasonal primary pollinator Hymenoptera *Genus* (*subgenus*), and secondary pollinators Diptera family Syrphidae and order Lepidoptera for sampling seasons 2016, 2017, and 2018 at the Roundup, MT field site.

	2016			2017			2018		
Genus (*subgenus*)	Enrolled	Not Enrolled	No Graze	Enrolled	Not Enrolled	No Graze	Enrolled	Not Enrolled	No Graze
Lasioglossum (Dialictus)	269	221	120	1057	757	236	205	106	68
Agapostemon	173	164	175	846	930	361	147	178	95
Eucera	130	71	14	723	509	133	83	76	121
Halictus	37	55	26	117	49	48	22	25	53
Macrotera	31	0	8	0	0	0	0	0	0
Anthophora	23	33	12	122	139	64	38	76	26
Andrena	12	11	4	180	340	68	109	114	94
Bombus	10	14	29	51	51	95	23	18	64
Diadasia	10	7	11	7	4	1	0	0	0
Osmia	5	15	13	28	30	17	17	7	23
Anthidium	4	2	4	3	3	0	0	0	0
Panurginus	4	0	8	7	1	0	0	0	0
Lasioglossum (s.str)	4	9	14	133	37	146	27	30	57
Lasioglossum (Sphecodogastra)	2	6	4	43	45	48	26	16	23
Megachile	2	1	4	14	4	2	0	0	0
Melissodes	1	1	0	1	0	0	0	1	0
Lasioglossum (Evylaeus)	1	6	16	17	4	12	7	3	9
Dufourea	1	0	4	0	1	2	0	0	0

Table 11. Continued

	2016			2017			2018		
Genus (*subgenus*)	Enrolled	Not Enrolled	No Graze	Enrolled	Not Enrolled	No Graze	Enrolled	Not Enrolled	No Graze
Apis	0	1	14	2	5	3	0	8	8
Nomada	0	0	2	6	7	8	2	3	3
Ashmeadiella	0	0	1	0	0	0	0	0	0
Coelioxys	0	1	1	0	0	0	0	0	0
Hoplitis	0	0	1	0	0	0	0	2	0
Calliopsis	0	1	0	0	0	0	0	0	0
Colletes	0	1	2	6	1	3	0	0	0
Hylaeus	0	0	6	0	0	0	0	0	0
Stelis	0	0	1	0	1	1	0	0	0
Protandrena	0	0	0	0	2	1	0	0	0
Sphecodes	0	0	0	3	0	1	1	2	1
Ceratina	0	0	0	1	0	0	0	0	0
Family									
Diptera									
Syrphidae	3	7	64	52	26	37	7	7	9
Order									
Lepidoptera	36	39	96	602	302	734	79	102	103

Vegetation Cover

In 2017 at the Roundup site, trapping location grass, forb, flowering forb and prickly pear did not differ. Shrub cover was greatest in non-enrolled when compared to enrolled (*t* = 2.66; df = 2, 6; *p* = 0.04) and non-grazed (*t* = 3.68; df = 2, 6; *p* = 0.01). Bare ground was greatest in enrolled (*t* = 2.63; df = 2, 6; *p* = 0.04) and non-enrolled (*t* = 2.23; df = 2, 6; *p* = 0.05) when compared to non-grazed. Litter coverage was greatest in non-grazed when compared to non-enrolled (*t* = 3.38; df = 2, 6; *p* = 0.01) and enrolled (*t* = 3.34; df = 2, 6; *p* = 0.02). Lichen coverage was greatest in non-grazed (*t* = 3.31; df = 2, 6; *p* = 0.02) and enrolled (*t* = 2.72; df = 2, 6; *p* = 0.03) when compared to non-enrolled pastures (Table 12).

In 2018 at the Roundup site, trapping location shrub, forb, flowering forb, bare ground, litter, lichen, and prickly pear did not differ. Grass coverage was greatest (*t* = 2.72; df = 2, 6; *p* = 0.03) in non-grazed when compared to non-enrolled; however, non-grazed did not differ from enrolled which did not differ from non-enrolled (Table 12).

Table 12. Weekly least squared mean comparisons of vegetation metrics for sampling seasons 2016, 2017, and 2018 at the Roundup, MT field site. Letter groupings beside least squared means signify differences between the mean.

	Grass	Shrub	Forb	Flowering Forb	Bare Ground	Litter	Lichen	Prickly Pear
2016[1]	-	-	-	-	-	-	-	-
Enrolled	-	-	-	-	-	-	-	-
Non-Enrolled	-	-	-	-	-	-	-	-
Non-Grazed	-	-	-	-	-	-	-	-
2017								
Enrolled	14.08	15.33 b	7.58	4.31	25.31 a	19.14 b	4.69 a	3.06
Non-Enrolled	11.03	22.27 a	7.06	4.28	23.25 a	18.94 b	2.64 b	2.50
Non-Grazed	10.19	12.69 b	10.81	4.31	11.86 b	35.17 a	5.14 a	2.64
SE	1.14	1.84	1.21	0.68	3.61	3.39	0.53	0.19
F-value	3.21	7.21	2.82	0.00	4.03	7.55	6.22	2.44
df	2, 6	2, 6	2, 6	2, 6	2, 6	2, 6	2, 6	2, 6
P-value	0.11	0.03	0.14	0.99	0.08	0.02	0.03	0.17
2018								
Enrolled	5.8 ab	16.00	2.67	7.63	3.50	4.01	7.73	2.50
Non-Enrolled	2.83 b	17.76	3.17	9.70	4.47	3.13	11.03	2.83
Non-Grazed	6.87 a	12.60	2.50	8.43	3.50	3.76	7.17	2.50
SE	1.03	3.58	0.38	1.60	0.80	0.64	1.84	0.16
F-value	4.15	0.54	0.82	0.42	0.48	0.50	1.29	1.50
df	2, 6	2, 6	2, 6	2, 6	2, 6	2, 6	2, 6	2, 6
P-value	0.07	0.61	0.48	0.67	0.64	0.63	0.34	0.29

[1] 2016 vegetation metrics under Fish, Wildlife, and Parks management. Vegetation sampling changed for the remaining two years.

Averaged weekly over three years of study, trapping location cover of grass, forb, flowering forb, lichen and prickly pear measurements did not differ among treatments. Shrub coverage was greatest (t = 3.26; df = 2,6; $p = 0.02$) in non-enrolled when compared to non-grazed; however, non-enrolled did not differ from enrolled which did not differ from non-grazed. Bare ground was greatest (t = 2.58; df = 2,6; $p = 0.04$) in enrolled when compared to non-grazed; however, enrolled did not differ from non-enrolled which did not differ (t = 2.34; df = 2,6; $p = 0.06$) from non-grazed. Although we did not statistically capture a difference, a *p*-value of 0.06 is indicating ecological significance.

Litter coverage was greatest (t = 2.41; df = 2,6; *p* = 0.05) in non-grazed when compared to non-enrolled; however, non-grazed did not differ (t = 2.28; df = 2,6; *p* = 0.06). Although we did not statistically capture a *p*-value of 0.06, it is indicating ecological significance. Finally, enrolled did not differ from non-enrolled (Table 13).

Table 13. Averaged weekly least squared mean comparisons of vegetation metrics across two study years 2017 and 2018 at the Roundup, MT field site. Letter groupings beside least squared means signify differences between the mean.

	Grass	Shrub	Forb	Flowering Forb	Bare ground	Litter	Lichen	Prickly Pear
Enrolled	10.32	15.64 ab	5.35	5.82	15.39 a	12.26 ab	6.08	2.80
Non-Enrolled	7.30	20.23 a	5.29	6.74	14.71 ab	11.76 b	6.45	2.65
Non-Grazed	8.68	12.65 b	7.03	6.18	8.06 b	20.89 a	6.06	2.58
SE	0.98	1.64	0.86	0.90	2.01	2.68	0.89	0.12
F-value	2.36	5.40	1.33	0.27	4.06	3.68	0.06	0.87
df	2, 6	2, 6	2, 6	2, 6	2, 6	2, 6	2, 6	2, 6
P-value	0.18	0.05	0.33	0.77	0.08	0.09	0.94	0.46

Discussion

To my knowledge, this study is one of the first to address the effects of livestock grazing on pollinator communities within Montana rangelands. Over three years of study at the Roundup study site, I found that the greatest activity-density of pollinators was within enrolled and non-enrolled pastures when compared to non-grazed pastures. Additionally, Hymenoptera pollinators were also greatest within the enrolled and non-enrolled pastures over the non-grazed. However, secondary pollinators were greatest on non-grazed when compared to non-enrolled.

For sampling year 2016, my results indicate that there were more total pollinators on the enrolled and non-enrolled pastures. Secondary pollinators showed no difference among all three treatments. Sampling year 2017 has similar results demonstrating the greatest total pollinator activity-densities were within enrolled and non-enrolled pastures. However, secondary pollinator activity-density was greatest in enrolled and non-grazed pastures. Sampling year 2018 results do not indicate differences between treatment activity-densities. Vegetation data for 2016 were rather slim. During the 2016 field season, vegetation protocol called for one initial vegetation cover reading via Daubenmire frame at the beginning of each sampling season. This later changed in 2017 and 2018 to include a weekly Daubenmire estimate of vegetation cover metrics at each sampling location. Therefore, 2017 and 2018 vegetation cover data at Roundup were much more thorough and 2017 vegetation data presented much variability. All metrics except flowering forbs were significant in some capacity and showed great variability between pastures. Vegetation cover for 2018 differed within the grass cover class and all other cover classes of shrub, forb, flowering forb, bare ground, litter, lichen, and prickly pear did not show any significance. While Roundup may present a wide diversity of plant-insect interactions, my results demonstrate the importance of geographic location including vegetation and precipitation and changes between pastures when analyzing pollinator communities.

All samples in this study were collected from pastures that were either 1) non-grazed and part of active cattle ranches that were not-enrolled in the USDA NRCS SGI rest-rotation program; 2) deferred from grazing and part of active cattle ranches that were enrolled in the USDA NRCS SGI rest-rotation program; and 3) long term idle and a part of the LMWR. Given the nature of this sampling, this indicates and suggests that all data represents metrics of post-treatment effects. As suggested by a recent report, the SGI within the same sampling locations, measuring the effects of grazing can exhibit a ‘lag’ effect and may also be influenced by weather and other confounding variables (FWP, 2016). While there was no direct impact of grazing during sampling, livestock grazing can impact pollinators through a different, however critical alternative pathway: vegetation. Livestock grazing may alter the richness of a flower community and the floral resources utilized by pollinators (Carvell, 2002; Mayer *et al*., 2006; Sjödin *et al*., 2008; Kimoto *et al*., 2012; Lazaro *et al*., 2016). Pollinators tend to track the vegetation communities as floral resources are necessary food sources (Potts *et al*., 2003; Elwell *et al*., 2016) and as such, vegetation tends to be an important predictor for pollinator abundance within livestock grazed pastures (Carvell, 2002; Sjödin, 2007; Kearns & Olivera, 2009; Lazaro *et al*., 2016). However, floral communities respond in diverse, often complex ways to grazing (Elwell *et al*., 2016) varying in intensity, and geographic location (Vulliamy *et al*., 2006; Yoshihara *et al*., 2008). Extant literature documents the variety of flowering plant responses to grazing as negatively (Kimoto *et al*., 2012; Tadey *et al*., 2015), positively (Carvell, 2002), or treatment dependent (Sjodin, 2006; Elwell *et al*., 2016). Much of these responses are a result of plant adaptations to grazing; those that are adapted with a history of grazing likely respond positively to grazing and negatively when the plants are generally undisturbed (Elwell *et al*., 2016).

As forbs are the most important nectar resources for pollinators, it would be rather intuitive to suggest that the differences we are seeing within our pollinator collections are correlated with forb cover. However, this is not what my results suggest and differences between cover classes are apparent within our Roundup study site pastures. In 2017, bare ground resources were greatest in the enrolled and non-enrolled pastures. In contrast, forb and flowering forb resources did not differ between enrolled, non-enrolled, and non-grazed pastures. This suggests that the differences I am seeing in primary and secondary pollinator activity-densities in 2017 are associated with bare ground and litter cover.

With a majority of native bees nesting within the ground or decaying vegetation cavities (Black *et al*., 2011; Wilson & Carril, 2016), the availability of bare ground for potential nesting sites is pertinent. *Lasioglossum* (*Dialictus*) and *Agapostemon*, both sweat bees in the Halictidae family, are two *Genera* of bees that were consistently trapped through all three treatments and in all three years.

Lasioglossum (*Dialictus*) is a small bee between 3 and 8 mm long. The *Genera Lasioglossum* contains about 1,800 species globally that range in sociality from solitary to primitively eusocial (Wilson & Carril, 2016). With such a large number of species, this bee is common within North America and usually found in abundance (Wilson & Carril, 2016). Due to the smaller size of this *Genera* of bee, they generally have short flight distances (Waddington, 1979) and as described in a study by Kimoto *et al*., did not appear to be sensitive to livestock grazing (2012). They suggest that this *Genus*, *Lasioglossum*, may not be sensitive to grazing due to their limited flight distances to adjacent fields with potentially more resources. Kimoto *et al*. also note that corresponding literature suggest sweat bees have a preference toward compacted soils and bare ground for nesting sites (Potts & Willmer, 1997, 1998; Kimoto *et al*., 2012). *Agapostemon* are another common native bee to Northern America. This *Genera* of bee is much larger than *Lasioglossum*, and thus is capable of longer flight distances to retrieve floral food resources. *Agapostemon*, like *Lasioglossum*, nests within the ground and may nest in communally (Wilson & Carril, 2016), keeping the entrance free of debris

(Abrams & Eickwort, 1981; NRCS, 2009). While there is no division of labor, *Agapostemon spp*. may share nest entrances to reduce the risk of their nests being parasitized by cuckoo bees (Wilson & Carril, 2016). It has also been documented that species of the *Genera Agapostemon*, attempt to maximize their floral rewards to flight energy costs by making direct flights to denser patches of flowers (Waddington, 1979). It is well noted in literature on native pollinators and livestock grazing that in addition to directly cutting away at floral resources utilized by native pollinators, livestock may also directly trample pollinators or nesting sites, reduce soil stability, and compress soils (Gess & Gess, 1983; Sugden, 1985; Potts *et al*., 2005; Vulliamy *et al*., 2006; Williams *et al*., 2010; Kimoto *et al*., 2012). While in the case of the sweat bee and many other native bees, soil compression and availability of bare ground may be a positive to their nesting behaviors (Wilson & Carril, 2016).

Eucera is another bee that was commonly collected within multiple treatments over the course of all three study years. Like the two above, *Eucera* is a ground nesting bee and prefers sandy or clayey soils, flat areas, and build a small mound around the entrance (Wilson & Carril, 2016). *Eucera* are a commonly and abundantly found bee of North America within the Eucerini tribe and often commonly found in the early spring (Wilson & Carril, 2016). While many of the species within this *Genera* are generalist, there are some specialists on specific floral resources (Wilson & Carril, 2016).

While not commonly collected, but noteworthy is the *Genus Macrotera*. This *Genera* is commonly found within the southwestern states but has been documented up through the pacific north coast. However, to my knowledge, this is the first time *Macrotera* has been documented within central Montana as it was ultimately trapped in

2016 within the enrolled pastures at the Roundup site and within the top five most collected for this site and year. This *Genus* of native bee consists of 30 known species- most of which are specialists, and all are known to nest within the ground (Wilson & Carril, 2016).

Literature suggests that most ground nesting bees prefer well-draining sandy to gravelly soils (NRCS, 2016), open areas, and generations generally return to the same nesting area (Michener, 2007). While some bees can cross significant landscapes, due to bees' keen flight abilities, they often stay in the same geographical area (Michener, 2007). In the 2007 book *Bees of the World* the author notes that disturbed areas often have more bees than undisturbed; however, this does not mean a lot of disturbance is better for bees (Michener, 2007). This follows the intermediate disturbance hypothesis that describes peak pollinator diversity at intermediate levels of grazing disturbance (Lazaro *et al*., 2016). This may explain, in part, why I am seeing higher activity-densities within the pastures that do experience cattle grazing part of the year when compared to the upper Lake Mason Wildlife refuge unit in which has not experienced grazing in at least seven years. However, other factors that impact the activity-densities that we recorded are elevation, precipitation, and temperature shifts year to year.

Precipitation and temperature affect native bees in two primary ways. The first is directly through their ability and timing of flight and foraging, and the second indirectly through its impact on the availability of or phenological stages of floral resources (Lawson & Rands, 2019). One report suggests that as the mean global temperature increases, precipitation will increase (Collins *et al*., 2013). As a result of increased precipitation, native bees and their floral resources are faced with another challenge associated with their native habitats. However, insects are equipped with integumal pores that line the surface of their body. These pores allow them to detect changes in barometric pressure and obtain the ability to ‘predict’ precipitation events (Lawson & Rands, 2019). When native bees detect a change, they often alter or increase their foraging before the precipitation event, continue to forage through light rain, or stop their foraging altogether (Lawson & Rands, 2019). Flight during precipitation events can be dangerous to the bees as impact with raindrops, or increased winds, could lead to lagged flight or even death (Lawson & Rands, 2019). On the other hand, precipitation and temperature alter the phenological stages of plants. Warmer temperatures decrease dormant stages within overwintering plants and growth, or flowering may become altered (Gray & Brady, 2016). This becomes problematic with native bee pollinators as some species have coevolutionary emergence times with their floral resources (Johnson & Anderson, 2010) and shifts in temperature and precipitation may consequently alter this. At the Roundup location, the non-grazed pastures were at a significantly higher elevation than the enrolled and non-enrolled pastures. Non-grazed pastures may have an effect of cooler temperatures and varied precipitation that will additionally impact or alter plant growth cycles or available floral resources. Sampling year 2016 had the lowest annual precipitation followed by 2017 and 2018 had the highest annual precipitation with 509 mm. The Roundup, MT 15 SW weather station reports a 1981-2010 annual average precipitation to be 361.95 mm. This means 2016 and 2017 are under average from 10 mm to approximately 50 mm and 2018 is over average by approximately 150 mm. While 2016 and 2017 are relatively dry years when compared to the average, we can begin to understand some more of the vegetation results I am seeing. Bare ground and litter showed no differences in 2018. While 2017 is a wetter year than 2016, it is still under average annual precipitation. The lower precipitation in 2017 may have impacted the growth and recovery of new plants the following spring in 2018. This could be a reason as to why I am recording no differences in 2018 bare ground and litter cover.

CHAPTER FOUR

IMPLICATIONS

With less than 1% of native prairies in their precolonial state remaining (NPS, 2015), the need to identify, research and summarize the working parts of these ecosystems should be a top priority of public and private land managers in the U.S. Plant communities of our native prairie ecosystems that are comprised of grasses, flowering forbs, shrubs, and legumes rely, in to varying degrees, on healthy and robust populations of pollinators (Reed, 1993). Moreover, contributing to approximately 80% of crop pollination and 35% of wild flowering plant pollination (Allsop *et al*., 2008) insect pollinators benefit many diverse biomes, including the sagebrush ecosystems (Dumroese, 2016). Bees represent the most important group of pollinators (Danforth *et al.*, 2013), and to date only 4,000 species have been documented in the United States (Gilgert & Vaughan, 2011; Hoffman, 2011). Recently the first bee, the Rusty Patch Bumble bee, was placed on the endangered species list in 2017 (Christopher, 2016 USFW publication). Despite this recent listing and the fact that arthropods represent over 70% of the documented animal species (IUCN, 2016), arthropods are one of the more underrepresented groups of animals when land management and conservation practices are considered (Winfree, 2010; Titley *et al*., 2017).

Through habitat fragmentation, destruction, and the disappearance of critical flowering resources, the implications that may occur if we ignore the fragility of pollinating insect species far surpass the challenges that we may face trying to conserve these organisms. As such, with landowners' goals for the land in mind and within the scope of this project, it appears that there is no single recommendation for pollination conservation in this region. The role of rotational grazing in these two biomes have opposing impacts and varies by bee *Genera*. In biomes similar to my Roundup site, intermediate levels of rotational grazing could benefit a large majority of native pollinators (Rambo & Faeth, 1999; Vulliamy *et al*., 2006; Yan *et al*., 2015) in this region, while having minimal impacts on the *Genera* of bees that negatively respond to cattle grazing. At Sidney, it is hard to say what kind of grazing method would benefit pollinators as our results demonstrate a neutral response. This is not to say, however, that various other cattle grazing, or land management techniques are not of use to pollinator conservation in other regions; the conservation goals should wholly be determined on the needs of the specified area and organisms present (Stanturf *et al*., 2014; Dumroese *et al*., 2015). Native pollinator conservation is not only important for the inherent value of the organism, but because of the value and many roles it takes on within an ecosystem that include crop pollination (Klein *et al*., 2007; Allsop *et al.*, 2008; Stein *et al*., 2017), wild flower pollination (Allsop *et al*., 2008; Koh *et al*., 2016; Stein *et al.*, 2017), food sources for other wildlife (Gilgert & Vaughan, 2011), and ecosystem maintenance through nutrient cycling (Kearns *et al*., 1998; Black *et al*., 2011).

Given the geographic location of the study sites within Montana and the long evolutionary history of the western U.S. being grazed by bison (Mack & Thompson, 1982), the ecosystems and vegetation that remain here are, to some extent, adapted to grazing disturbances in soils with high nutrients (Frank *et al*., 2016). Many studies highlight the positive influence rest-rotational cattle grazing systems can have on biodiversity (Carvell, 2002; Sayre *et al*., 2012; Enri *et al*., 2017; Lyons *et al*., 2017). One study even highlights improved abundance and species richness of flower-visiting insects under a rotational grazing system (Enri *et al*., 2017). When addressing the conservation needs of pollinators in this region, determining the needs and goals of the land without sacrificing the integrity of the existing native plants and animals (Krausman *et al*., 2009) coupled with the continued inclusion of conservation easements on private lands and open involvement on public lands should prove beneficial for pollinator conservation. Federal and state agencies should enhance pollinator habitat by maintaining a heterogenic landscape that provides ample resources for a diversity of insects (Öckinger & Smith, 2007; Delaney *et al*., 2015) and reduce high intensity disturbances to the land during peak flowering season (June-July) (Enri *et al*., 2017). The implementation of these conservation strategies, along with others, to increase or maintain the diversity and abundance of pollinating insects have shown to increase abundance and richness of bees, and even increase locally uncommon species (Enri *et al*., 2017).

In summary, many tools, including livestock grazing, are available for land managers to implement for conservation of native pollinators. However, any one tool used alone will not fully address the real needs of pollinator conservation if efforts to minimize anthropogenic disturbances, which reduce wild bee richness and abundance, are not implemented (Winfree *et al*., 2009). Regardless of whether these tools of pollinator conservation are used on private or public lands, they do allow for the flexibility needed to maintain or even enhance pollinator habitat and subsequent populations.

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APPENDICES

APPENDIX A

SUPPLEMENTAL POLLINATOR DATA

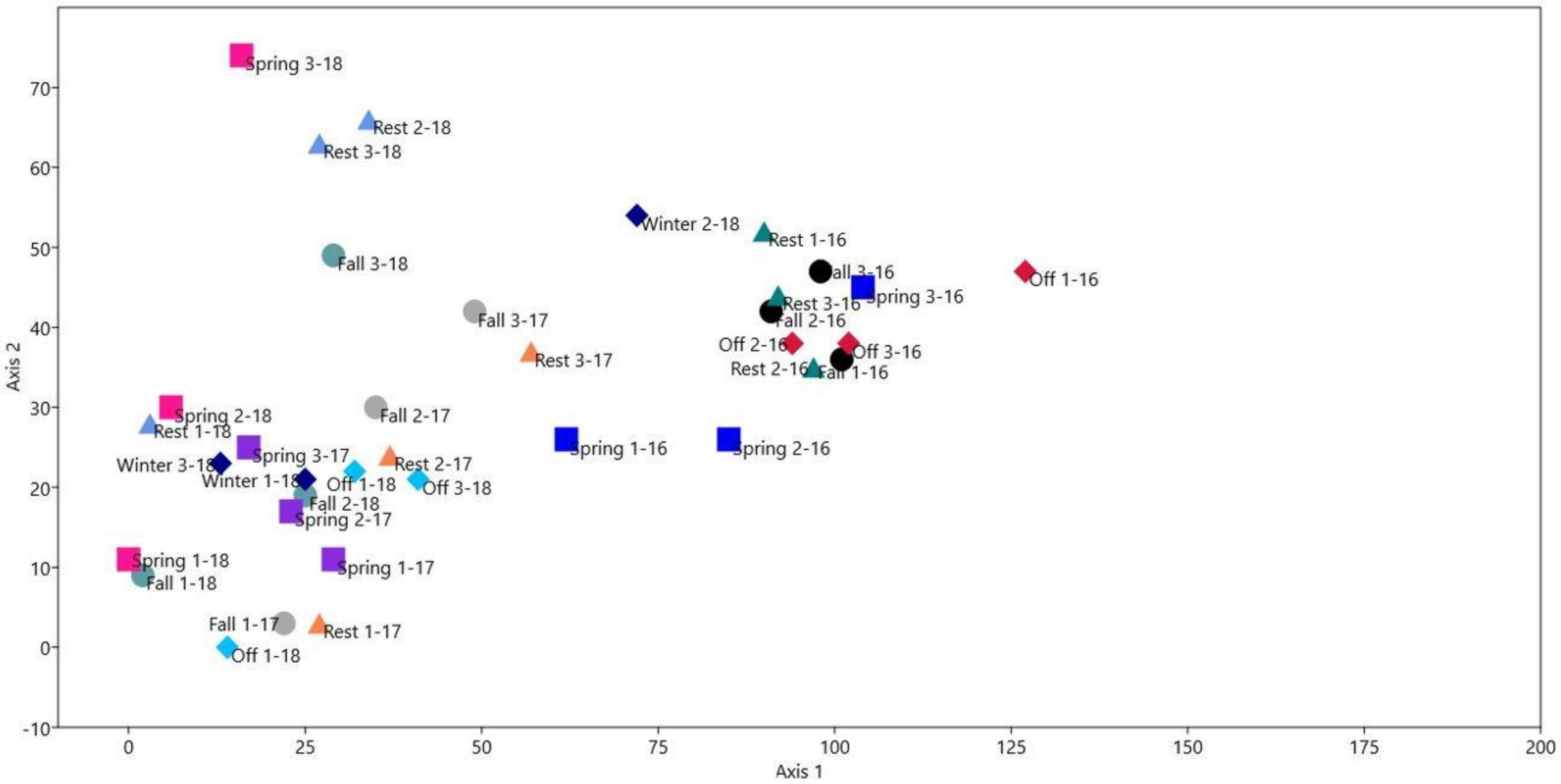

Figure A1. Detrended correspondence analysis of *Genus* level diversity at each of the Sidney, MT field sites across three years of study.

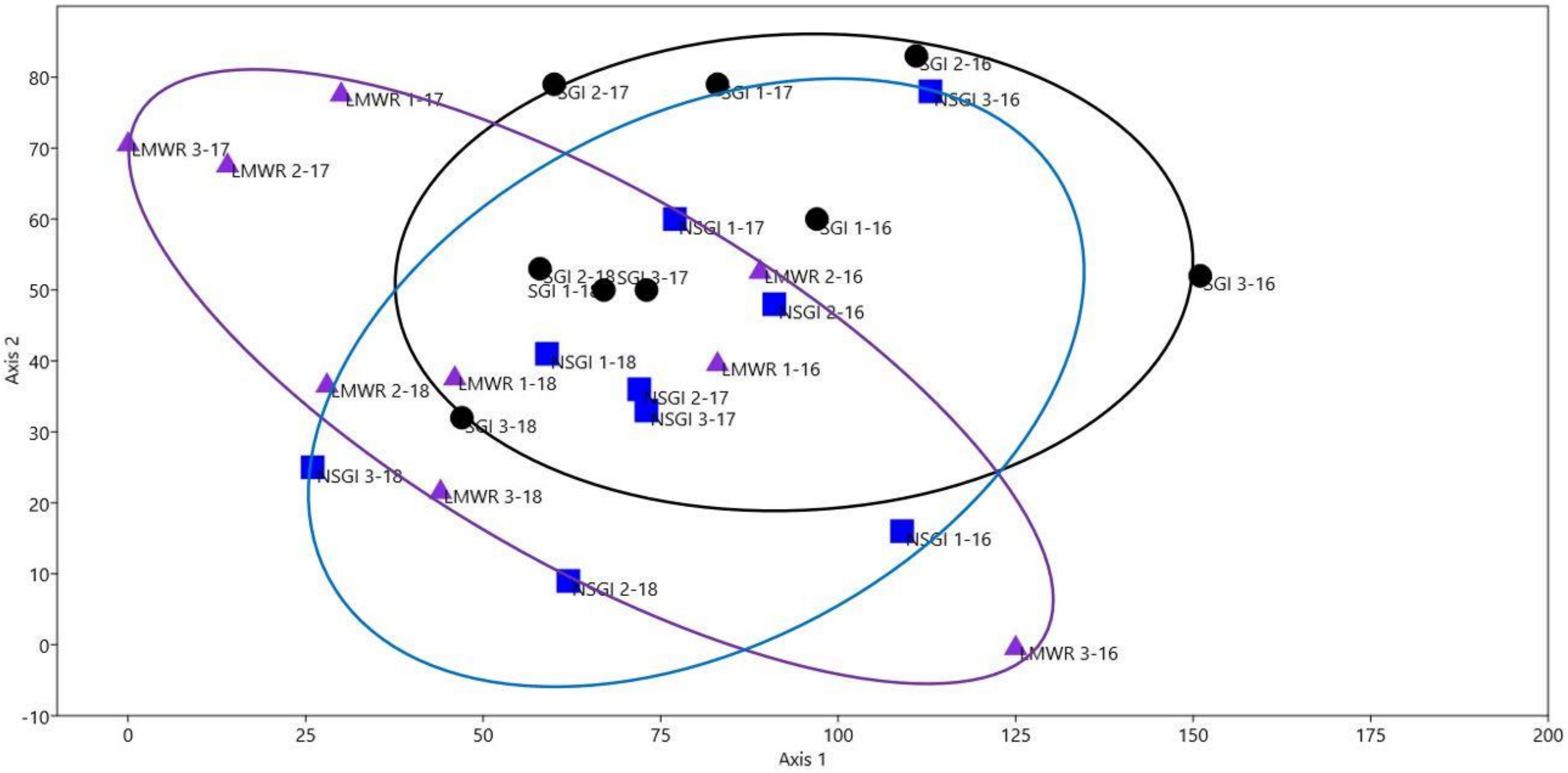

Figure A2. Detrended correspondence analysis of *Genus* level diversity at each of the Roundup, MT field sites across three years of study.

Table A1. Top 10 *Genus (subgenus)* recorded from each of the Sidney, MT field site in year 2016[1,2].

Fall 2016	Spring 2016	Rest 2016	Off 2016
L. (Dialictus)- 613	*L. (Dialictus)*- 234	*L. (Dialictus)*- 569	*L. (Dialictus)*- 418
Agapostemon- 207	*Agapostemon*- 91	*Agapostemon*- 167	*Agapostemon*- 138
Halictus- 115	*Halictus*- 72	*Halictus*- 109	*Halictus*- 64
Melissodes- 34	*L. (s.str)*- 9	*Bombus*- 33	*Melissodes*- 45
L. (s.str)- 30	*Melissodes*- 8	*L. (s.str)*- 30	*L. (s.str)*- 26
Bombus- 15	*Bombus*- 7	*Melissodes*- 24	*Andrena*- 9
Diadasia- 12	*Diadasia*- 6	*Diadasia*- 13	*Bombus*- 6
Andrena- 9	*Eucera*- 4	*Andrena*- 9	*Diadasia*- 6
Colletes- 7	*Protandrena*- 3	*Colletes*- 5	*L. (Sphecodogastra)*- 5
L. (Sphecodogastra)- 6	*Andrena*- 2	*L. (Sphecodogastra)*- 3	*Protandrena*- 3
	L. (Sphecodogastra)- 2	*Anthophora*- 3	
	Osmia- 2		

[1] *L.* denotes the *Genus Lasioglossum.*

[2] Winter 2016 pollinators were only keyed to Family and thus not included in this table.

Table A2. Top 10 *Genus (subgenus)* recorded from each of the Sidney, MT field site in year 2017[1,2].

Fall 2017	Spring 2017	Rest 2017
L. (Dialictus)- 1161	*L. (Dialictus)*- 869	*L. (Dialictus)*- 1446
Agapostemon- 1048	*Agapostemon*- 737	*Agapostemon*- 822
Halictus- 294	*Halictus*- 208	*Halictus*- 164
L. (s.str)- 185	*Eucera*- 140	*L. (s.str)*- 130
Andrena- 95	*L. (s.str)*- 95	*Eucera*- 96
Eucera- 85	*Andrena*- 78	*Andrena*- 87
Bombus- 58	*Bombus*- 74	*Bombus*- 60
Anthophora- 14	*L. (Sphecodogastra)*- 11	*L. (Sphecodogastra)*- 18
L. (Sphecodogastra)- 13	*Osmia*- 10	*Osmia*- 11
L. (Evylaeus)- 12	*Protandrena*- 8	*Melissodes*- 10
	Nomada- 8	

[1] *L.* denotes the *Genus Lasioglossum.*

[2] Off-easement and Winter pastures were not sampled in 2017 and thus not included in this table.

Table A3. Top 10 *Genus (subgenus)* recorded from each of the Sidney, MT field site in year 2018[1].

Fall 2018	Spring 2018	Rest 2018	Off 2018	Winter 2018
Agapostemon- 220	*Agapostemon*- 176	*Agapostemon*- 169	*L. (Dialictus)*- 238	*Agapostemon*- 18
L. (Dialictus)- 144	*L. (Dialictus)*- 158	*L. (Dialictus)*- 142	*Agapostemon*- 229	*L. (Dialictus)*- 14
L. (s.str)- 32	*Bombus*- 144	*L. (s.str)*- 57	*Bombus*- 31	*Bombus*- 5
Andrena- 22	*Andrena*- 41	*Bombus*- 41	*Eucera*- 27	*Halictus*- 4
Halictus- 20	*L. (s.str)*- 23	*Andrena*- 27	*Halictus*- 18	*L. (s.str)*- 3
Bombus- 16	*Eucera*- 21	*Halictus*- 21	*L. (s.str)*- 18	*Eucera*- 2
Eucera- 11	*Halictus*- 13	*Eucera*- 11	*Andrena*- 9	*Andrena*- 1
L. (Sphecodogastra)- 4	*L. (Sphecodogastra)*- 5	*L. (Sphecodogastra)*- 7	*Anthophora*- 5	
Megachile- 1	*Apis*- 3	*L. (Evylaeus)*- 7	*L. (Sphecodogastra)*- 4	
Nomada- 1	*L. (Evylaeus)*- 1	*Osmia*- 3	*Megachile*- 1	
Apis- 1			*Osmia*- 1	
			Apis- 1	

[1] *L.* denotes the *Genus Lasioglossum.*

Table A4. Top 10 *Genus (subgenus)* recorded from each treatment at the Roundup, MT field site in year 2016[1].

Enrolled 2016	Non- Enrolled 2016	Non- Grazed 2016
L. (Dialictus)- 269	*L. (Dialictus)*- 221	*Agapostemon*- 175
Agapostemon- 173	*Agapostemon*- 164	*L. (Dialictus)*- 120
Eucera- 130	*Eucera*- 71	*Bombus*- 29
Halictus- 37	*Halictus*- 55	*Halictus*- 26
Macrotera- 31	*Anthophora*- 33	*L. (Evylaeus)*- 16
Anthophora- 23	*Osmia*- 15	*Eucera*- 14
Andrena- 12	*Bombus*- 14	*L. (s.str)*- 14
Bombus- 10	*Andrena*- 11	*Apis*- 14
Diadasia- 10	*L. (s.str)*- 9	*Osmia*- 13
Osmia- 5	*Diadasia*- 7	*Anthophora*- 12

[1] *L.* denotes the *Genus Lasioglossum.*

Table A5. Top 10 *Genus (subgenus)* recorded from each treatment at the Roundup, MT field site in year 2017[1].

Enrolled 2017	Non- Enrolled 2017	Non- Grazed 2017
L. (Dialictus)- 1057	*Agapostemon*- 930	*Agapostemon*- 930
Agapostemon- 846	*L. (Dialictus)*- 757	*L. (Dialictus)*- 236
Eucera- 723	*Eucera*- 509	*L.(s.str)*- 146
Andrena- 180	*Andrena*- 340	*Eucera*- 133
L. (s.str)- 133	*Anthophora*- 139	*Bombus*- 95
Anthophora- 122	*Bombus*- 51	*Andrena*- 68
Halictus- 117	*Halictus*- 49	*Anthophora*- 64
Bombus- 51	*L. (Sphecodogastra)*- 45	*Halictus*- 48
L. (Sphecodogastra)- 43	*L. (s.str)*- 37	*L. (Sphecodogastra)*- 48
Osmia- 28	*Osmia*- 30	*Osmia*- 17

[1] *L.* denotes the *Genus Lasioglossum.*

Table A6. Top 10 *Genus (subgenus)* recorded from each treatment at the Roundup, MT field site in year 2018[1].

Enrolled 2018	Non- Enrolled 2018	Non- Grazed 2018
L. (Dialictus)- 205	*Agapostemon*- 178	*Eucera*- 121
Agapostemon- 147	*Andrena*- 114	*Agapostemon*- 95
Andrena- 109	*L. (Dialictus)*- 106	*Andrena*- 94
Eucera- 83	*Eucera*- 76	*L. (Dialictus)*- 68
Anthophora- 38	*Anthophora*- 76	*Bombus*- 64
L. (s.str)- 27	*L.(s.str)*- 30	*L. (s.str)*- 57
L. (Sphecodogastra)- 26	*Halictus*- 25	*Halictus*- 53
Bombus- 23	*Bombus*- 18	*Anthophora*- 26
Halictus- 22	*L. (Sphecodogastra)*- 16	*Osmia*- 23
Osmia- 17	*Apis*- 8	*L. (Sphecodogastra)*- 23

[1] *L.* denotes the *Genus Lasioglossum.*

APPENDIX B

ENVIRONMENTAL PARAMETERS

Table B1. Weather variables collected from the Sidney-Richland KSDY METAR weather station located at 47.7000 and -104.2000 (USpest.org).

	2016	2017	2018
Temp. (°C)	7.94	6.76	5.37
Wind (kph)	15.08	15.3	13.44
Rain (mm)	706.37	9.14	8.128

Table B2. Weather variables collected from the Horsethief, MT HORM8 RAWS weather station located at 46.4256 and -108.6742 (USpest.org).

	2016	2017	2018
Temp. (°C)	9.26	8.56	7.12
Wind (kph)	7.69	7.27	6.39
Rain (mm)	313.44	351.28	509.02

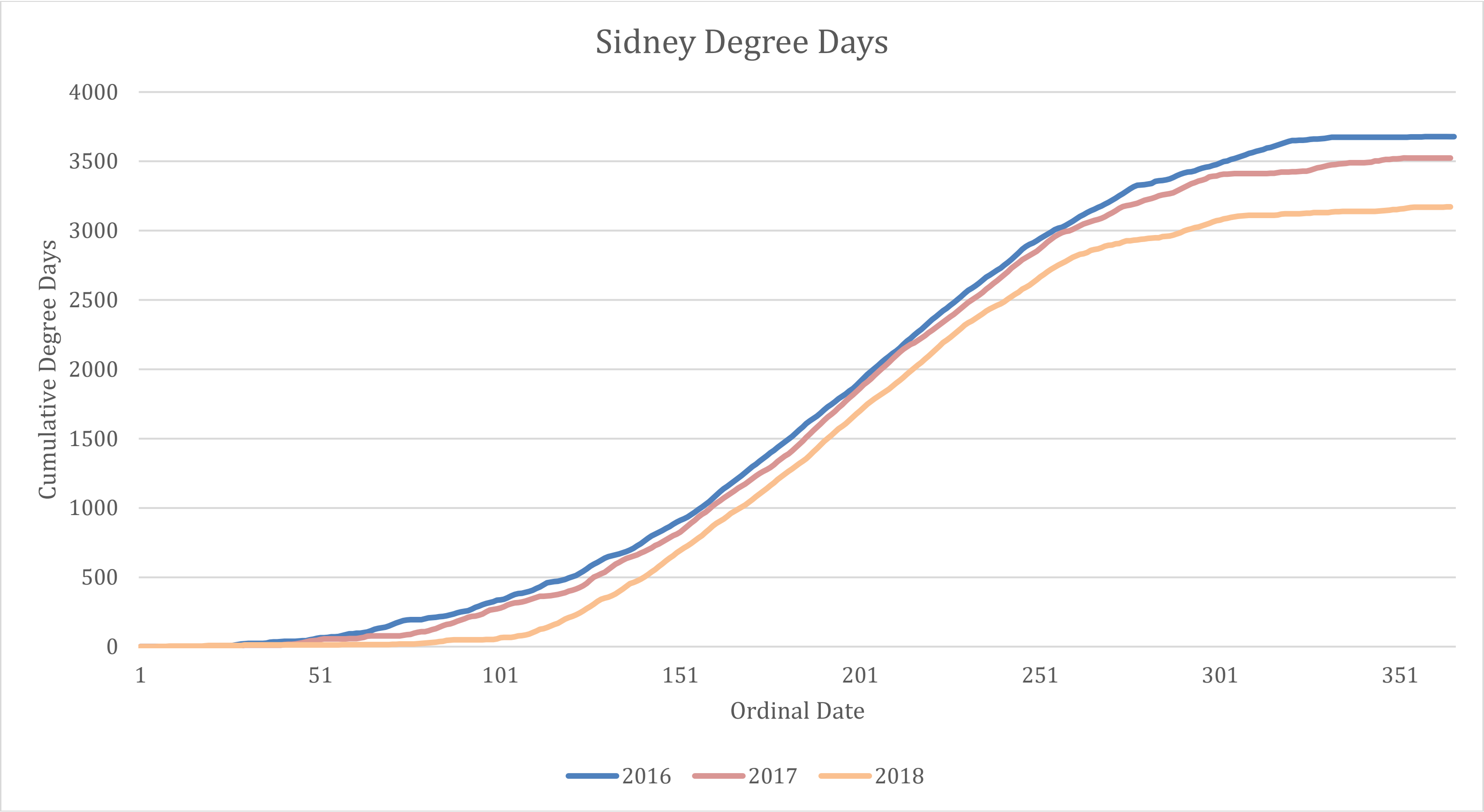

Figure B1. Cumulative degree days retrieved from the Sidney-Richland KSDY METAR weather station located at 47.7000 and -104.2000 (USpest.org). 2016 sampling began at about 166-degree days, followed by 138 in 2017, and 135 in 2018.

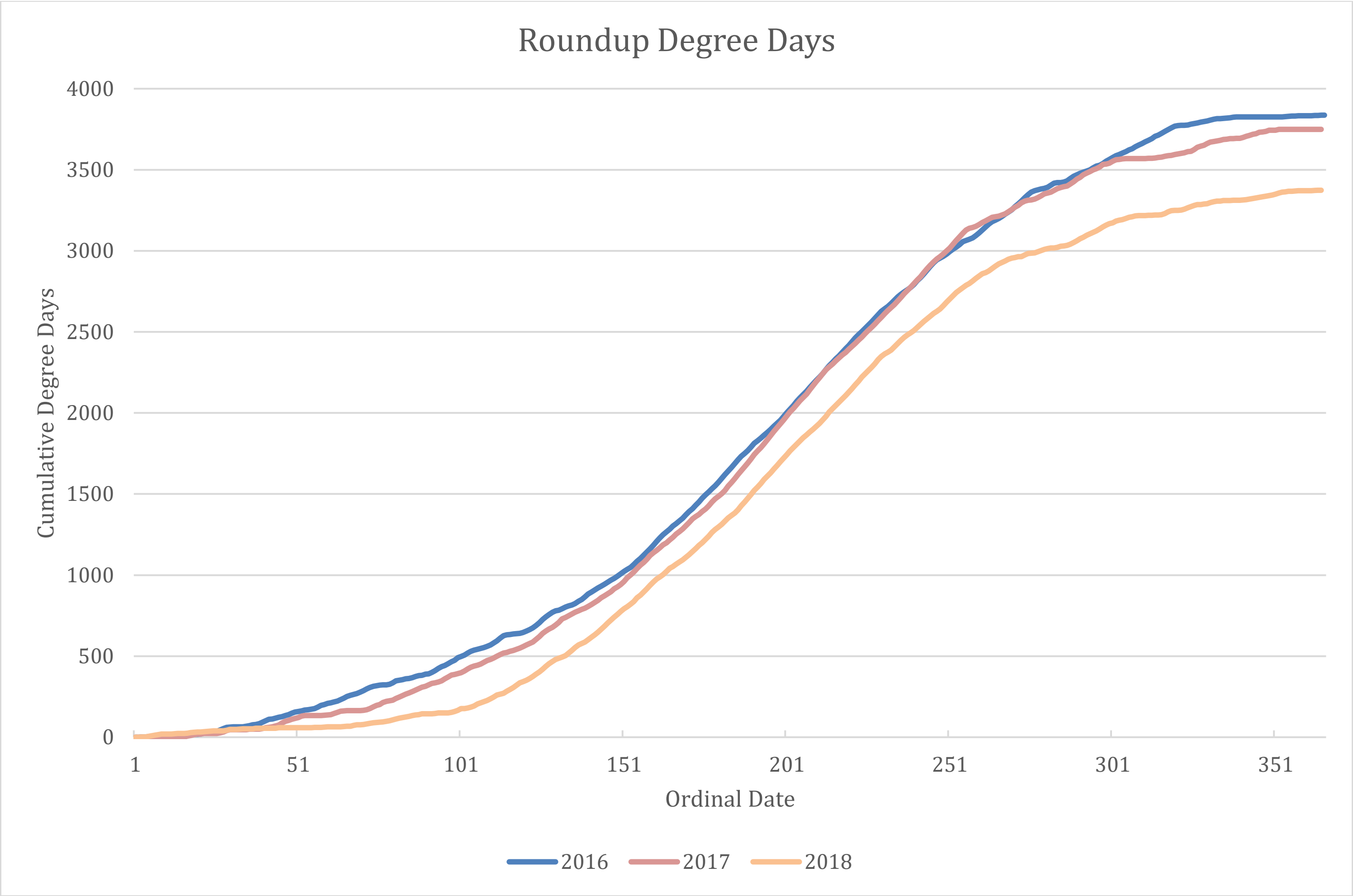

Figure B2. Cumulative degree days retrieved from the Horsethief, MT HORM8 RAWS weather station located at 46.4256 and -108.6742 (USpest.org). 2016 sampling began at about 159-degree days, followed by 139 for 2017, and 136 for 2018.