

Katrina Di Bacco
Dr. Buddle
April 2, 2017

The effects of forest management on pollinator biodiversity and community structure, in
Montebello, QC

Abstract

Pollinators provide essential ecosystem services for agricultural landscapes as well as the reproduction of wild flowers in forested habitats. Forestry practices rapidly alter natural ecosystems and can thus be a potential threat to already declining wild pollinator populations. This project aims to determine if three forestry management practices, old growth, strip cutting and clear cutting impact pollinator diversity and their community structure, as well as aims to compare collection methods and observe general patterns in pollinating communities. Pollinators were collected for 10 weeks in the summer of 2016, in an area of Montebello, QC, that was recently impacted by forestry. The two methods used to collect pollinators were collecting flower visitors and setting out yellow pan traps. These insects were then sorted by order and three groups of insects beetles (Coleoptera), true bugs (Hemiptera) and bees (Hymenoptera:Apoidea) were identified. The diversity of the three management techniques was analyzed using four diversity indices, species richness, Shannon index, Simpson index and inverse Berger-Parker index. The community structure was plotted using an NMDS ordination plot and plant-pollinator interactions were observed with interaction plots. In general, the diversity among the three forest management sites was not found to be significantly different, except for the Simpson diversity index which found a difference in diversity between old growth communities and strip cut communities. The community composition was also found to be significantly different indicating that species between forest management techniques are generally unique to that habitat. There were more plant-pollinator interactions in the strip cut and clear cut sites than in old growth sites, however all three sites showed evidence that non-native flower species were visited by the most number of pollinator species. Unlike past studies which show that forestry positively impacts the diversity of bees, this study suggests there is no difference in the pollinator diversity after forestry. However, this study highlights the importance of habitat heterogeneity in forested communities as different groups of pollinators rely on resources found in clear cut, strip cut and old growth habitats.

Introduction

Insect pollination is of the most valuable services that ecosystems provide as it is crucial for the reproduction of wild flowers and responsible for over 75% of crops consumed by humans (Allsopp et al. 2008, Cariveau et al. 2013, Mace et al. 2012). In natural ecosystems such as forests and meadows, insects pollinate over 78% of flowers in temperate zones and 94% of flowers in the tropics (Ollerton et al. 2011). Understory herbaceous plants, the main diet items of herbivores, are most dependent on insect pollinators in these forested habitats (Gilgert et al. 2011). Additionally, wild pollinator declines often cause a reduction of wild flower diversity as fewer pollinators correlate with fewer plants reproducing (Biesmeijer et al. 2006). A diverse set of pollinators also pollinate plants more efficiently as some pollinators are preferentially attracted to a certain plant species (Memmott 1999, Loweinstein et al. 2015).

Insect pollinators are generally classified as domestic pollinators or wild pollinators. Domesticated pollinators, such as the European honey bee (*Apis mellifera*) are highly managed by humans in artificial hives and are

generally used to pollinate agricultural crops (Aizen and Harder 2009). Contrarily, wild pollinators are a diverse group that are not managed by humans (Faegri & Van Der Pijl 2013). The most efficient wild pollinators are bees (Order: Hymenoptera: Apoidea) which are then followed by the butterflies and moths (Order: Lepidoptera) (Gilgert et al. 2011). Flies (Order: Diptera) are also considered important wild pollinators since they are the most abundant of groups, but they are less efficient than bees (Jauker et al. 2012, Fishbein and Venable 1996). Beetles (Order: Coleoptera) are thought to be the first insect pollinators, but are also generally not highly efficient (Proctor and Yeo 1996, Thien et al. 2000). Finally, some true bugs (Order: Hemiptera) are also considered potential pollinators as they visit flowers regularly to consume pollen and mate, but of all orders previously mentioned they are the least efficient pollinating species (Albano et al. 2009, Ishida et al. 2009, Wheeler 2001).

In the past 20 years, there has been a growing concern for the fate both wild and domestic pollinators as bee keepers, farmers and researchers have noticed a decline in both abundance and species richness (Aizen and Harder 2009, Henry et al. 2012, Potts et al. 2010). Although one definitive cause in a decrease in pollinators has not been discovered it is suspected that many sub-lethal causes interact which then cause a decline in population health and productivity (Rundlöf et al. 2015, Tylianakis 2013). The north-east of the United States has observed up to a 50 % reduction in wild bee species in the last 120 years (Burkle et al. 2013). The continuing decline of pollinators could significantly diminish the ecosystem services provided which are raising serious concerns for both global food security and maintenance of forested habitats (Dicks et al. 2013).

Habitat loss and habitat modification are suspected as a potential driver in most cases of wild pollinator declines (Klauser and Peduzzi 2007). When a virgin habitat is converted to agricultural or urban landscapes, the abundance and species richness of wild flowers decreases which subsequently leaves pollinating insects with reduced food availability. Modification of

habitats can also destroy nests or limit viable nesting sites which halts reproduction of the insects (Tylianakis 2013, Potts et al. 2010).

The forestry industry contributes to substantial habitat modification in Canada as approximately 100 000 km² of forests are cut down each year due to increasing demands for wood products (Natural Resources Canada 2016). Due to the obvious losses in biodiversity, the forestry industry has been moving toward more sustainable techniques, in hopes to improve forest regeneration and protect the wildlife that rely on these habitats (Drever et al. 2006). In order to determine which techniques are least detrimental, impact assessments are completed before and after forests are harvested. Included in these assessments are often species of high concern, economic value or biological indicators of healthy ecosystems (Mace et al. 2012). Since pollination is an important service provided for both humans and for the maintenance of diverse ecosystems, pollinator communities should also be considered when determining sustainable forestry practices (Maleque et al. 2009).

This study focuses on how pollinating communities have been impacted by forestry techniques. The two forestry management techniques on the property, clear cutting and strip cutting will be compared with old growth forests which act as a control (Doyon et al. 2005, Thompson et al. 2014).

The clear cut technique is used when large areas of forest need to be removed. This method mimics large scale natural disturbance, since the forest floor is completely exposed to sunlight (Lencinas et al. 2014). In contrast, a strip cut management practice cuts forests in alternating strips leaving intact forest between cuts. The advantages of strip cutting are that the forest regenerates faster due to the buffer system alternating the deforested area (Graham-Sauvè et al. 2013). The habitat created by the two previously mentioned techniques vary greatly from old growth forests. Old growth forests have not been disturbed by forestry and are thus often used as a control treatment (Lencinas et al. 2014). Although they provide important habitats for many species, the vegetation in old growth forests is often dominated by large trees which shade the forest understory (Romey et al. 2007).

Due to the large shaded areas in old growth forests, pollinator abundance and diversity tend to be lesser than in open areas. These open areas are often able to hold more wildflowers as more sunlight is available. Additionally, open areas are often warmer which can contribute to a greater productivity for the insects. Both strip cuts and clear contain large open areas which both tend to have more wild flowers and warmer microclimates than shaded areas. These management techniques thus have been found to increase the abundance and species diversity of bees (Devoto et al. 2012, Steffan-Dewenter et al. 2002, Romet et al. 2007, Winfree et al. 2007). However, clear cut and strip cut sites tend to also have a great abundance of non-native flowers which may outcompete native flowers. Therefore, the pollinators will have an abundance of food due to the surge of non-native flowers, but may be hindering the natural succession of the forest (Hengstum et al. 2014, Thompson et al. 2014). In some cases, these non-native flowers in clear cut and strip cut areas may actually distract native bees from

pollinating native flowers and thus decrease the abundance of native flowers reproducing (Drever et al. 2006, Potts et al. 2010).

Additionally, strip cut and clear cut communities have less available wood debris for pollinator nesting sites than old growth habitats. The woody debris is especially important for solitary bees, beetles and true bugs as they need a moist woody environment to reproduce. Studies observing bee diversity have concluded that leaving remnant wood on the ground after forestry will help the wood nesting bees but will impair some wild flowers from growing (Steffan-Dewenter et al. 2002). However, studies on the impact of forestry on beetles and true bugs have yet to make any meaningful recommendations.

The objectives of this research are firstly, to compare pollinator diversity in each of the forest management technique in order to determine if one management type creates habitat that increases pollinator abundance and species richness. The second objective is to observe when the most common pollinator species are most abundant as this may provide insight to when pollinating communities are most vulnerable. The third objective is to compare the community structure and species composition in the three management techniques in order to determine if different pollinators are in all treatments. Lastly, the two collection methods will be compared in order to determine if one method is generally better at collecting the most species or most individuals. This study observed pollinators from the orders Coleoptera, Hemiptera and Hymenoptera: Apoidea. The purpose of this study is to ultimately examine if the forestry management techniques alter the pollinator communities at Kenauk Nature.

Methods

Sampling Design

The study was completed at Kenauk Nature, a forested property of 650 km², in Montebello, Quebec. The forests are generally categorized as mixed forests, although some areas of the property were previously seeded with various coniferous trees. The vast majority of deciduous forests are dominated by Sugar Maple (*Acer saccharum*), whereas the majority of coniferous stands are White Spruce (*Picea glauca*). This property has been logged in different areas, thus in order to determine which method of harvest was used and when, a map produced by CERFO (Centre d'enseignement et de recherche en foresterie de Sainte-Foy inc) was consulted. The map outlined areas of the property which have recently been clear cut, strip cut in 2010-2015 as well as areas that were never cut. Subsets of these forests were visited and three sites were chosen for each forestry method studied.

The geographic location and elevation (Figure 1, Appendix 1) were measured using a Garmin ETREX 20 GPS. The three clear cut sites were within the closest range of each other, as this was the only area of the property that had been completely cut down in 2015. Finally, the old growth areas were chosen by small openings in the canopy where flowers could grow. Since

these were the most difficult to find, the old growth sites were likely the least similar as their flower composition and canopy cover differed across the sites.



Figure 1. Map of the Kenauk Nature property. The clear cut sites are indicated in yellow and annotated with “CC”, the strip cut sites are indicated in blue and annotated with “SC” and the old growth sites are indicated in green and annotated with an “OG”. The elevation of each site is indicated after the site code and in appendix 1.

Insects were collected from June 6th 2016 until August 12th 2016, thus sampling took place for most of the 2016 summer season. The two methods used to capture insects are described below.

Pan trap method

When first arriving at a site, a rectangle of 4 meters by 2 meters was measured. Then 6 yellow SOLO bowls were placed on the outline of this rectangle at 2 meter intervals, which created 2 lines of pan traps that were 2 meters apart. Before a pan trap was placed on the ground, slits were punctured onto the sides of the trap, to eliminate excess water if it rained (Moreira et al.

2016, Romey et al. 2007). Additionally, the pan traps were punctured with 15 cm nails in order to secure them to the ground.

Each week, the pan traps were filled half way with soapy water. After 48 hours, the pan traps were collected and arthropods were strained from the water using a fine meshed cloth. The cloth and arthropods were placed in a plastic container with ethanol to temporarily preserve the specimens (Romey et al. 2007, Potts et al. 2010). Once the arthropods in all 54 pan traps were strained and preserved, the containers were emptied into a sorting tray and sorted according to taxonomic order. The orders accounted for were: Araneae, Hymenoptera, Hemiptera, Coleoptera, Diptera and Lepidoptera, all other specimens were classified as 'Other'. The sorted specimens were counted, then placed in 1.5mL micro-centrifuge tubes and submerged in 70% ethanol.

The specimens in the orders, Hymenoptera, Hemiptera, Coleoptera were identified to genus or species in the laboratory using a binocular stereoscopic microscope and insect keys (Romey et al. 2007).

Netting method

In addition to the pan trap method, the sites were visited for one hour of effort time on a dry day every week. Flowers were observed and any arthropod that came in contact with them was collected with a sweep net. This method is more specific to collecting pollinators and although it is more time consuming, it has been found to be more efficient than the pan trap method (Popic et al. 2013). Once an insect was collected, the date, site and species of flower it was found on was indicated on a label. The flower species were determined in the field using field guides or a specimen was picked and identified using Gleason and Cronquist (1963) key.

The collected insects were placed in vials with their label and stored in the freezer. Each week, these insects were counted and pinned (Romey et al. 2007). These insects in the orders, Hymenoptera, Hemiptera and Coleoptera were identified to genus or species in the laboratory using a binocular stereoscopic microscope and insect keys (Romey et al. 2007).

Data analyses

In order to compare pollinator diversity in the three types of forest management techniques, the species that were collected over 10 times in a management technique were plotted according to their rank. This comparison gives insight on the most abundant species of each management technique and if the most abundant species are similar across all techniques.

Four different diversity measures were then calculated. First, the species richness of each site was determined by counting the number of pollinating species found at each site. The number of species was then plotted on a boxplot in order to compare each management type (R Development Core Team 2016). Secondly, the Shannon diversity index was calculated for each site. The Shannon diversity index accounts for both the number of individuals collected as well

as the species richness of that site in form of entropy. Thus if a site has a higher diversity, it will also have a higher Shannon diversity index (Beck & Schwanghart 2010, Romey et al. 2007). The formula for the Shannon Diversity index is:

$$- \sum_{i=1}^S p_i \ln p_i$$

where p_i is the proportion of individuals belonging to a species (i) (Jost 2006). A boxplot demonstrating the differences in diversity among forest treatments was created in R (R Development Core Team 2016). The third diversity index calculated was the Simpson index. This Simpson index calculates the probability that two species collected will not belong to the same species. Therefore, if a site has a high Simpson index, it also has a high diversity as there is a lesser chance that many individuals all belong to one species. The formula for the Simpson index is:

$$1 - \sum_{i=1}^R p_i^2$$

where p_i is the proportional abundance of a given species (i) (Jost 2006).

Finally, the inverse Berger-Parker index was calculated which represents the inverse of proportional abundance of the most common species in a site. The Berger-Parker diversity index infers species evenness of a community since if the proportion of the most common species is high then we can infer that most individuals in a population will be the same species. Therefore, if the inverse of the Berger-Parker index is high, a community can be considered diverse as the most common species does not have a large proportional abundance. The formula for the inverse if the Berger-Parker index is:

$$\frac{N}{n_{max}}$$

where N is the total number of individuals and n_{max} is the abundance of the most common species (Beck & Schwanghart 2010). All four diversity indices were tested for significant difference between forest management technique with an ANOVA (R Development Core Team 2016).

In order to compare when the most common pollinator species were most abundant, the data was first split by the insects' order. Then the most common pollinating species were selected and the abundance sorted by collection date using a pivot table on EXCEL. A line graph was then made for each insect order which shows the abundance of a particular species for each collection week of summer 2016.

To observe community structure of the three management techniques, all the samples collected throughout the summer were put on one spreadsheet and using the function `aggregate()`, each site was condensed into one row (Oksanen et al. 2016). A non-metric multidimensional scaling (NMDS) ordination plot was then created using the function `metaMDS()` in R (R Development Core Team 2016). The forest management techniques were tested for significance via a PERMANOVA using the "vegan" package (Oksanen et al. 2016). To

compare the pollination interactions that have been observed, an interaction plot was constructed for each management technique using the function `plotweb()` which is contained in the package “bipartite” in R (Dormann et al. 2009).

The collection methods, the netting and pan traps, were compared by counting how many species were collected with each method as well as how many individuals were collected. This data was then sorted into management techniques so that a final histogram could be constructed in EXCEL, which compared both the management techniques and the collection methods (Potts et al. 2010).

In order to determine if adequate samples were collected to obtain meaningful results, a rarefaction curve was created for each site using the package “vegan”. A Chao1 asymptotic species richness was also calculated in order to estimate the number of Hemiptera, Coleoptera and Apoidea species that are likely in the entire property (Chao 1984, Oksanen et al. 2016).

Results

Throughout the 10 sampling weeks, 1510 individual insects (Coleoptera, Hemiptera and Hymenoptera: Apoidea) were collected which belong to 126 species or genera (Appendix 2). Of these insects, 1005 individuals were identified as potential pollinators which belong to 70 species or genera. The most common species collected was a Hemiptera nymph (*Podisus brevispinus*) which is not a pollinating species. The two next most common species, *Neokolla hieroglyphica* and *Graphocephala coccinea* are common leafhoppers that can also be found on flowers and thus potential pollinators (Overall and Rebek 2015).

When only considering pollinating species, the first two most common species are the leafhoppers *Neokolla hieroglyphica* and *Graphocephala coccinea*, although these are followed by two common bumble bee species *Bombus teranus* and *Bombus vagans* which are very efficient pollinators (Gilgert et al. 2011). When comparing the most abundant pollinating species at each forest management technique, the most common species in the clear cut sites was *Bombus ternarius* at 62 individuals (Figure 2a) whereas *Neokolla hieroglyphica* was the most common in both the strip cut and old growth sites at 73 and 83 individuals respectively (Figure 2b, Figure 2c). Additionally, both the old growth and strip cut sites had *Graphocephala coccinea* as their second most abundant species. *Neokolla hieroglyphica* is the only species which was collected at least 10 times in all forest management techniques (Figure 2a, Figure 2b, Figure 2c). Other overlapping species are the *Bombus vagans* and *Lasioglossum sp.* as they are found in both the clear cut and strip cut sites (Figure 2a, Figure 2b). The family of plant bugs (*Miridae sp.*) was abundant in both the old growth and clear cut sites (Figure 2a, Figure 2c). Generally, most of the abundant species differed by the management technique, how ever a few species were found in all techniques.

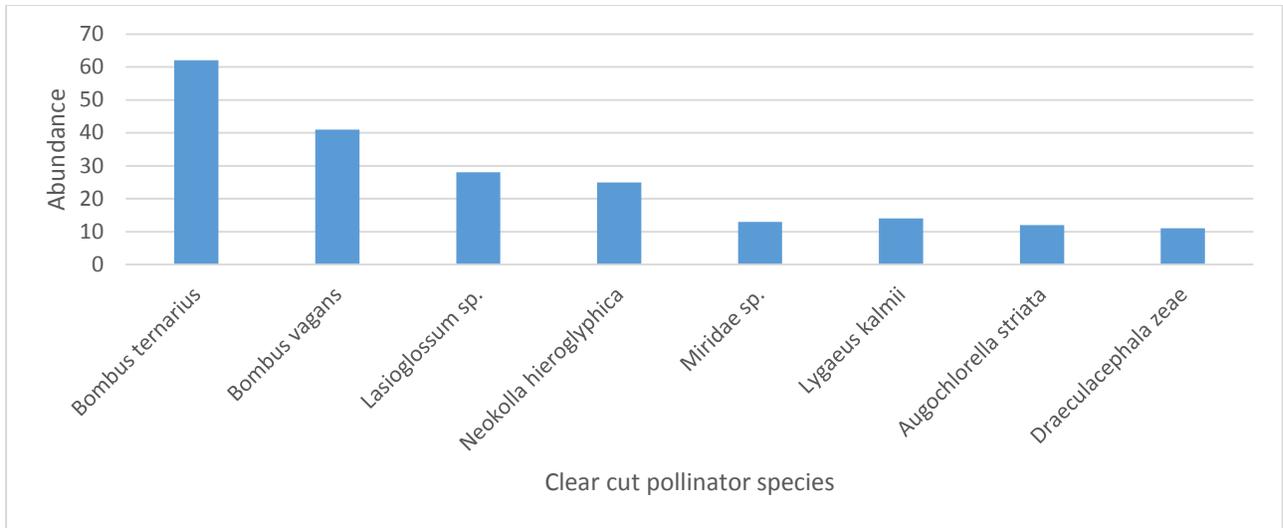


Figure 2a. Rank-abundance plots for species with greater than 10 individuals collected in the clear cut sites.

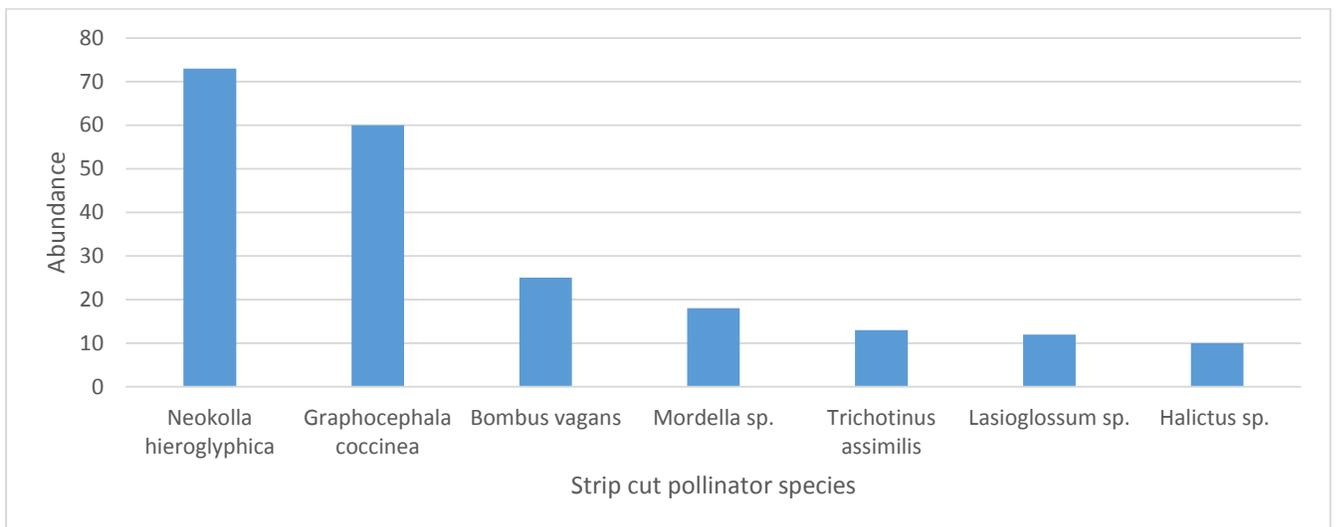


Figure 2b. Rank-abundance plots for species with greater than 10 individuals collected in the strip cut sites.

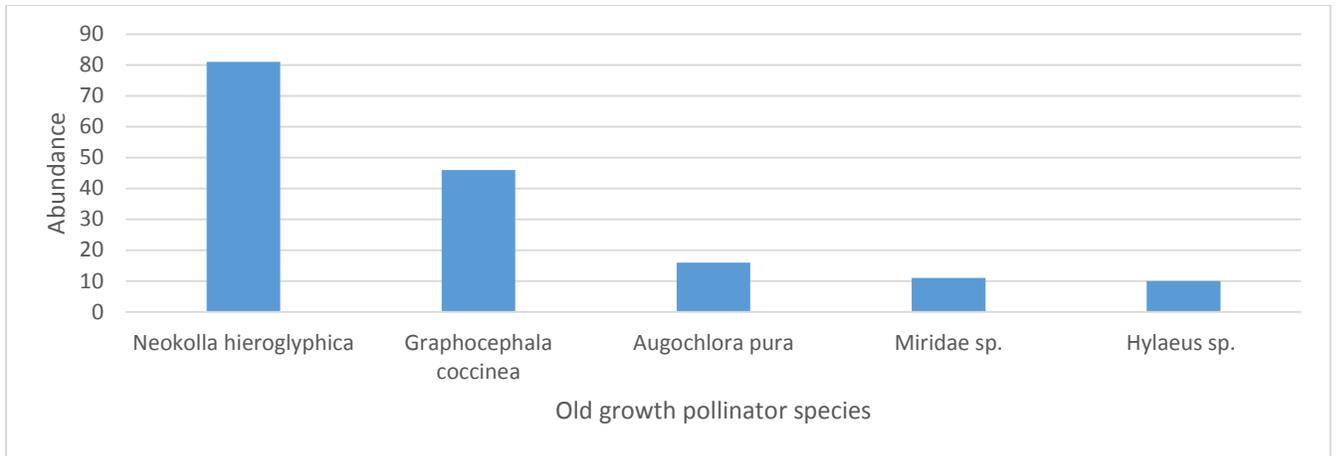


Figure 2c. Rank-abundance plots for species with greater than 10 individuals collected in the old growth sites.

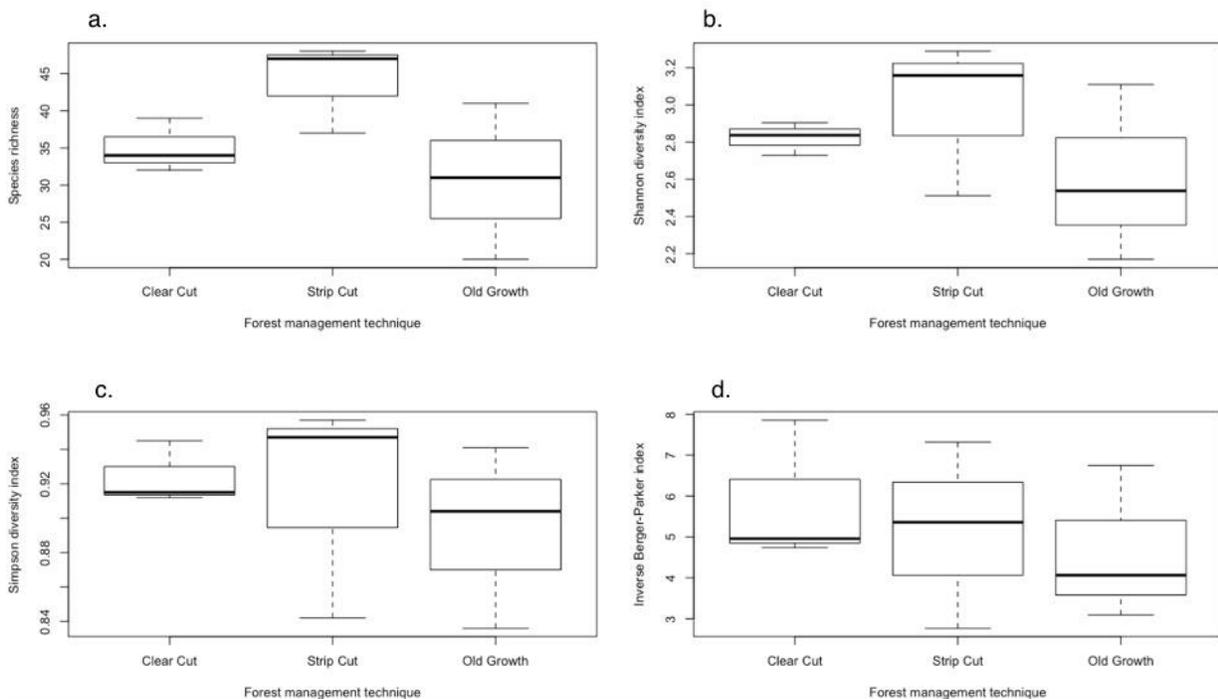


Figure 3. Four boxplots showing different diversity indices. Figure 3a. Species richness for each forest management technique. Figure 3b. Shannon diversity index, accounting for species richness and species evenness for each forest management technique. Figure 3c. Simpson diversity, indicating the probability that two individuals collected will not belong to the same species, for each forest management technique. Figure 3d. Inverse Berger-Parker index, the inverse of the proportion of the most common species in the community, for each forest management technique.

In the all four diversity indices, the strip cut sites had the highest median diversity, the clear cut sites had the second highest median diversity and the old growth sites had the lowest median diversity (Figure 3).

Throughout the entire summer 2016 season, the strip cuts also had the highest mean individuals caught (133) and highest mean species richness (44). The clear cuts and old growth sites had a similar mean number of individuals and mean species richness caught per site with 107 individuals and 35 species and 104 individuals and 30 species caught respectively (Figure 3a). The mean abundance and mean species richness were also not significantly different ($p>0.05$) among forest management treatments (Appendix 2).

The strip cuts sites also had the highest mean Shannon diversity index (2.986) with the clear cut sites having a similar mean diversity values (2.832). The old growth sites had both the most variation in diversity as well as the lowest Shannon diversity (2.605). However, when testing for statistical significance with an ANOVA, the differences in diversity were not significant across sites ($p>0.05$) (Figure 3b).

Although the strip cuts have the highest median Simpson diversity index, the clear cut sites actually have the highest mean diversity (0.924) (Figure 3c). The strip cuts have a similar mean diversity (0.915) whereas the old growth sites have the lowest mean Simpson diversity (0.890) (Appendix 2). The comparison in Simpson index with an ANOVA demonstrated there was a significant difference in Simpson diversity indices in the old growth and strip cut sites ($p=0.03$). The old growth and clear cut sites did not have significantly different Simpson diversity indices ($p>0.05$) and neither did the strip cut sites and clear cut sites ($p>0.05$).

The Berger-Parker index describes the proportional abundance of the most common species, thus a high inverse of the Berger-Parker index describes a more even and thus diverse community. Unlike with the other diversity indices, all the forest management techniques have quite large variations in Berger-Parker indices among sites. The clear cut sites have the highest mean inverse Berger-Parker index (5.85) whereas the strip cut sites had the second highest mean inverse Berger-Parker index (5.15) (Figure 3d). The old growth sites overall had the lowest inverse Berger-Parker index (4.64) indicating that the most common species made up a large proportion of the total abundance (Figure 3d, Appendix 2). However, when testing for statistical significance with an ANOVA, the differences in diversity were not significant across sites ($p>0.05$).

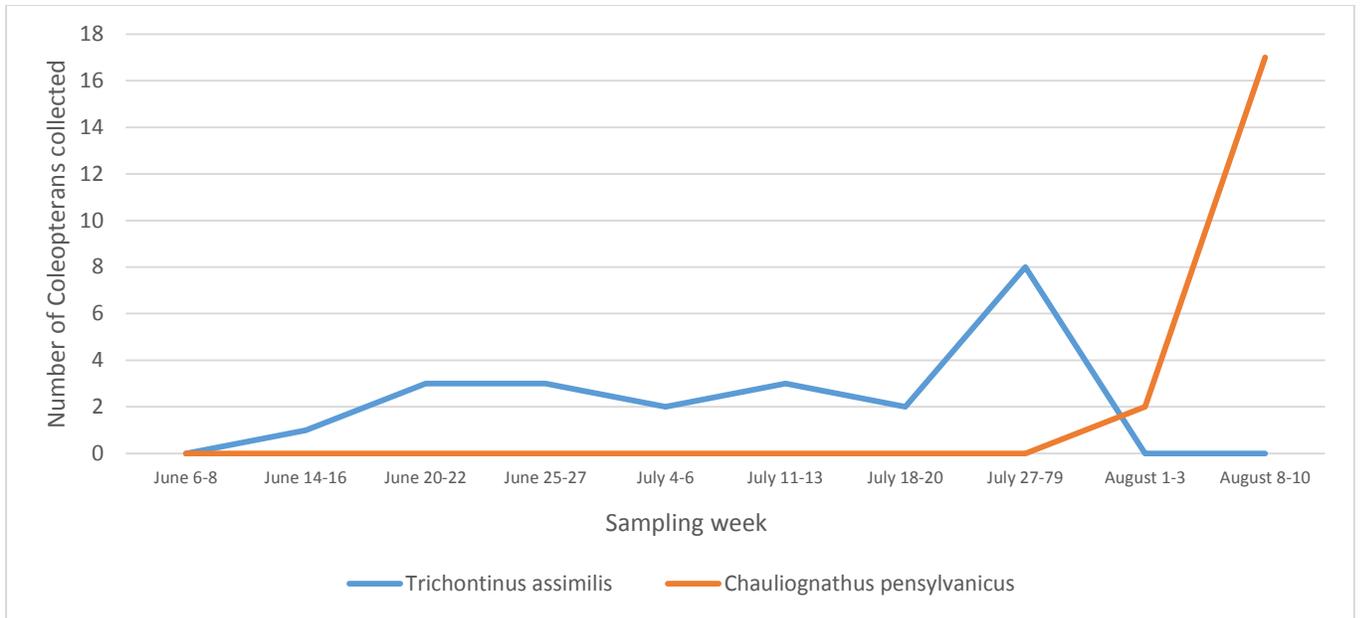


Figure 4a. Phenology of the two most abundant beetle species, *Trichontinus assimilis* and *Chauliognathus pensylvanicus*

The two most common beetle species, the Flower beetle (*Trichontinus assimilis*) and the Goldenrod soldier beetle (*Chauliognathus pensylvanicus*) did not occur at the same time during the summer season of 2016. The Flower beetle was collected on average 2 or 3 times a week however, during the week of July 27th there was a large increase as 8 individuals were collected. Flower beetle individuals were not collected during the last two weeks of sampling. However, the Goldenrod soldier beetle was only collected during the last two weeks of sampling indicating they may be only present in the late summers (Figure 4a).

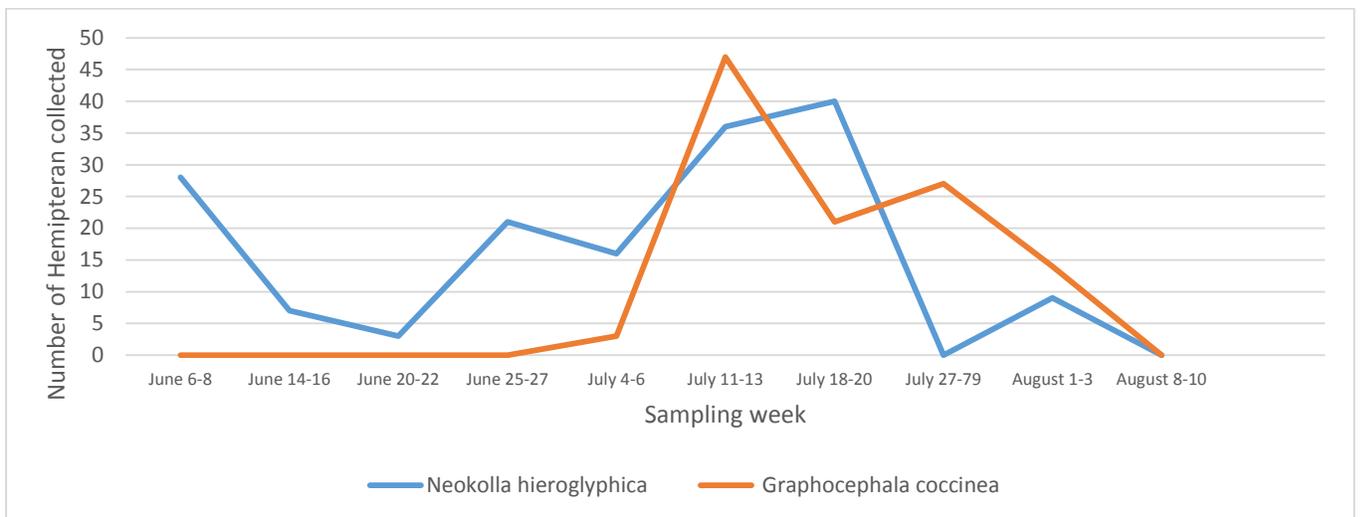


Figure 4b. Phenology of the two most abundant truebug species, *Neokolla hieroglyphica* and *Graphocephala coccinea*.

Both leafhopper species, *Neokolla hieroglyphica* and *Graphocephala coccinea* at first appear to follow a similar phenological patterns since their populations peak during the weeks of July 11 to July 18. However, *Neokolla hieroglyphica* population fluctuates throughout the entire sampling period, whereas *Graphocephala coccinea* only appears after July 4th and then its population declines after the week of July 27 (Figure 4b).

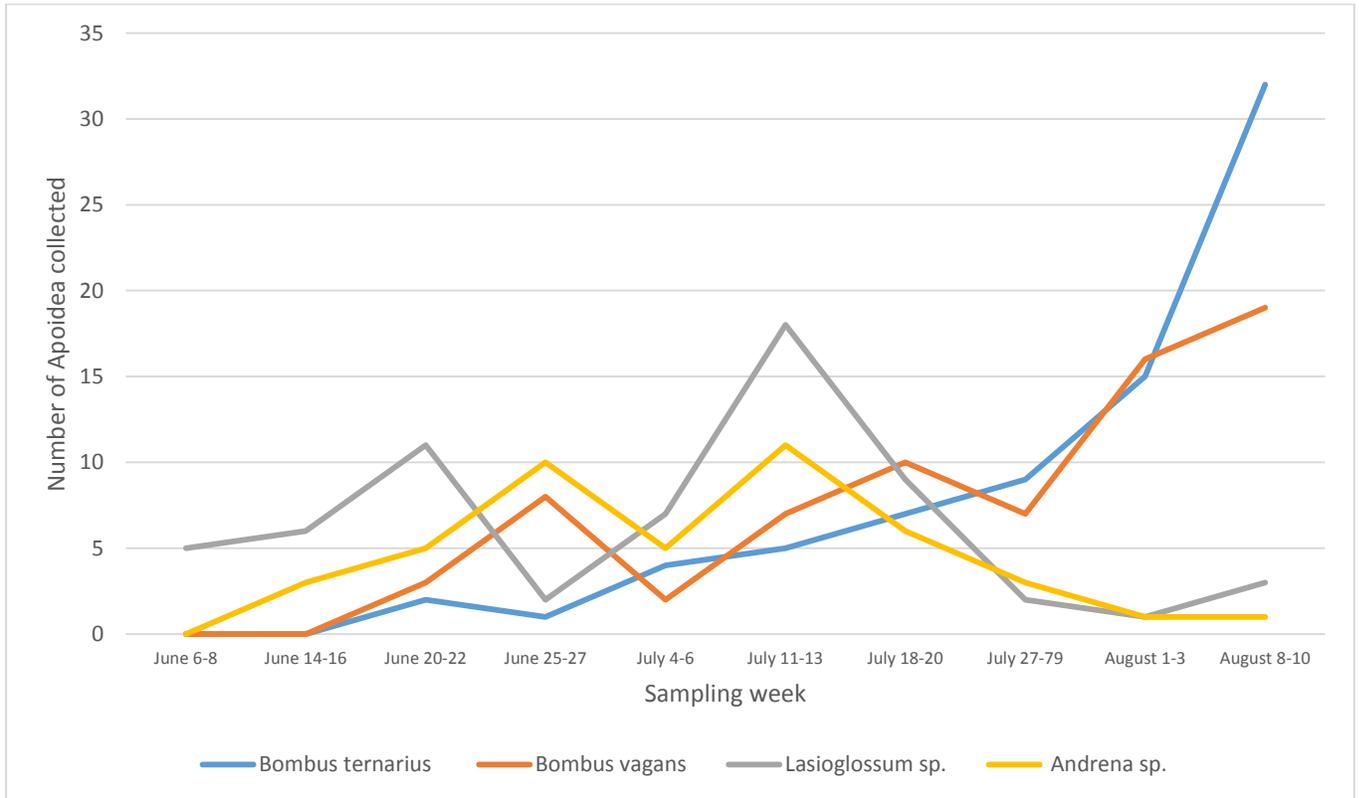


Figure 4c. Phenology of four most abundant bee species, *Bombus ternarius*, *Bombus vagans*, *Lasioglossum sp.* and *Andrena sp.*.

The four bee species collected, the Orange-belted bumblebee (*Bombus ternarius*), Half-black bumblebee (*Bombus vagans*), Sweat bee (*Lasioglossum sp.*) and the Mining bee (*Andrena sp.*) follow different phenological patterns throughout the summer season of 2016. The two bumblebee species increase in individuals collected each week and the maximum number of species collected occurred during the last week (Figure 4c). This trend may indicate that their populations are more abundant or more active in the late summer (Memmot et al. 2007). Contrarily, the two medium sized bee genera, the Sweat bee and Mining bees were collected frequently throughout the entire summer season yet, begin to decrease in numbers by the end of July (Figure 4c). This trend indicates that their population peaks in the middle of summer and they are either preparing to nest or reproducing by the end of the summer (Memmot et al. 2007).

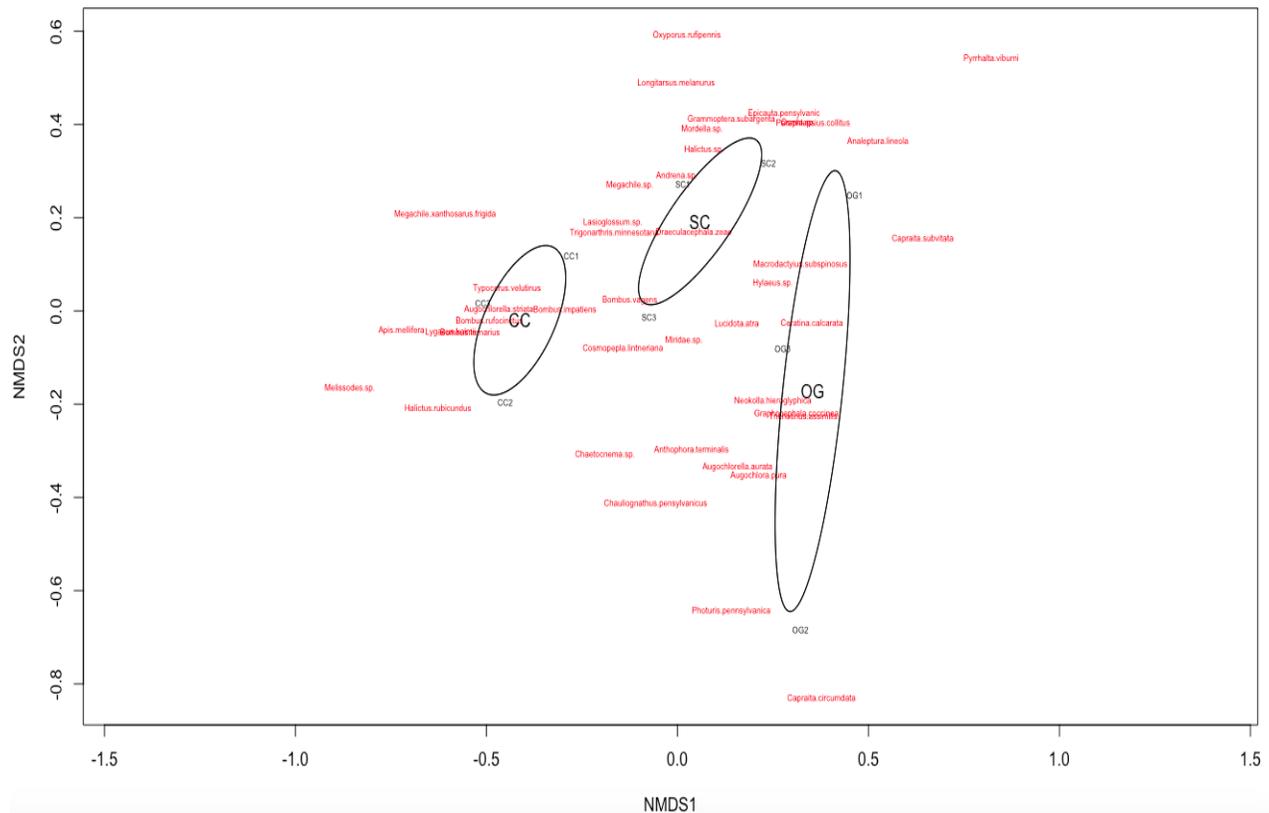


Figure 5. NMDS Ordination of pollinator communities with management types overlaid. The “CC” represents clear cut sites, the “SC” represents strip cut sites and the “OG” represents old growth sites.

The forest management techniques explain over half the variation in the pollinator community data ($r^2 = 0.63$). The ordination plot demonstrates that there is a significant difference ($P < 0.002$) in the community composition of pollinators across the three management techniques when tested with PERMANOVA. The species which are closer to the left of the first axis were generally found in more clear cut communities, whereas species near the right of the first axis are found more in old growth communities (Figure 5). The ellipses, representing a 95% confidence interval of each forest management technique, demonstrate that the clear cut and strip cut communities are clustered more tightly than the old growth communities indicating that there is more certainty in the communities that were forested (Figure 5). This is likely since the old growth sites were least similar to each other and thus had less consistent insect communities.

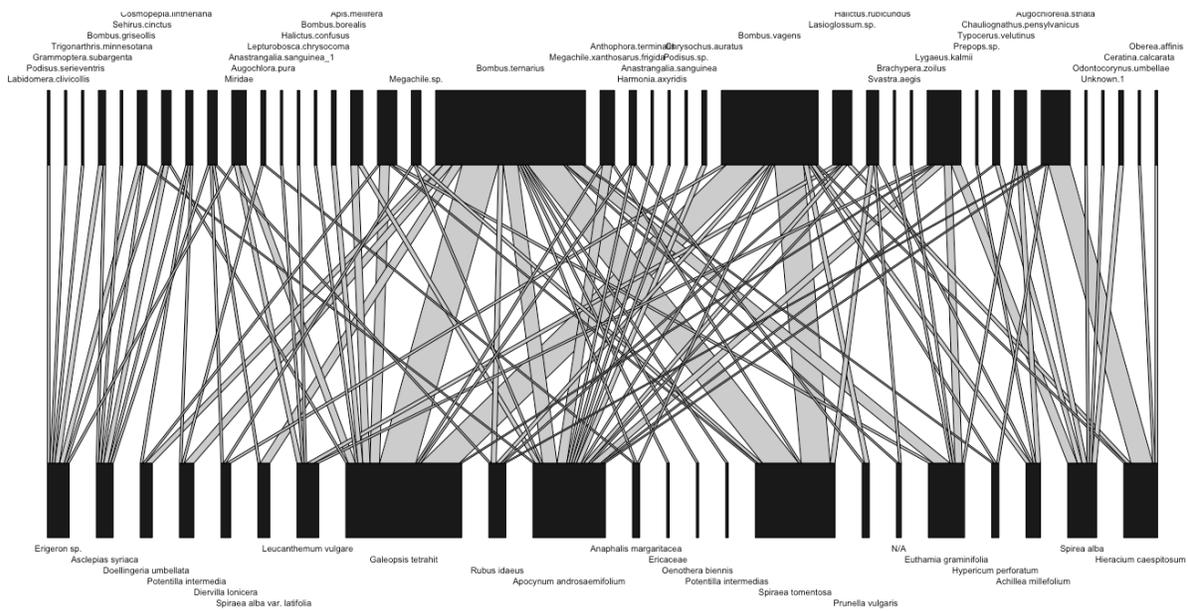


Figure 6a. Clear cut forest management interaction plot. The boxes at the bottom represent the flower species being pollinated whereas the boxes at the top represent the pollinator species collected with the netting method. The thicker interaction lines indicate that the interaction was observed more frequently.

There are many infrequent interactions that occur in the clear cut sites and only a few strong interactions. *Bombus ternarius* and *Bombus vagans* were both frequently recorded pollinating Hemp-nettle (*Galeopsis tetrahit*), a non-native and weedy mint species and Steeplebush (*Spiraea tomentosa*), a small native rose flower. One other strong interaction is *Augochlorella striata* pollinating Hawkweeds (*Hieracium caespitosum*). The flower which seems to have the most pollinators is Fly-trap dogbane (*Apocynum androsaemifolium*). Fly-trap dogbane is a short native bush with bright and fragrant flowers. Additionally, the pollinator which interacted with most flowers was *Bombus ternarius*, however this bumblebee was also the most abundant in the clear cut sites (Figure 6a).

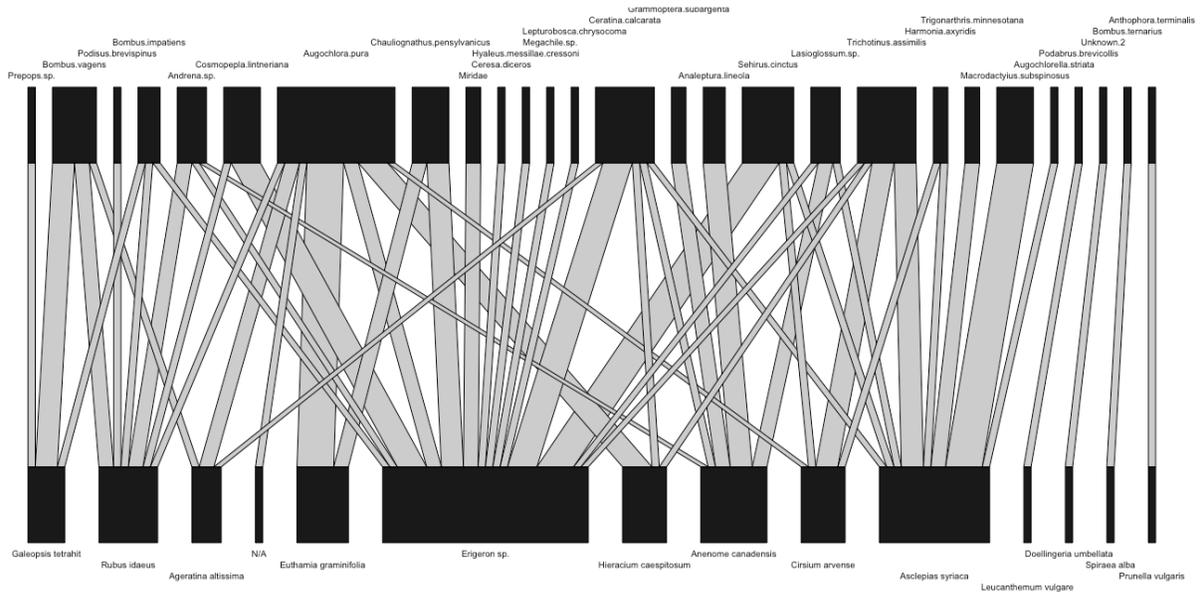


Figure 6c. Old growth forest interaction plot. The boxes at the bottom represent the flower species being pollinated whereas the boxes at the top represent the pollinator species collected with the netting method. The thicker interaction lines indicate that interaction was observed more frequently.

The old growth sites overall had the fewest interactions (Figure 6c) as they also had the fewest number of individuals collected with nets (Figure 7b). The strongest interactions were *Augochlora pura* with Grass-leaved goldenrod (*Euthamia graminifolia*) and *Macroductylus subspinosus* with common Milkweed (*Asclepias syriaca*). The flower most commonly pollinated was Fleabane (*Erigeron sp.*) (Figure 6c). Fleabane is similar to the oxeye daisy in that it is a non-native weedy species, and thus may also act as a pollen reserve for some species, but also outcompete native wild flowers (Thompson et al. 2014).

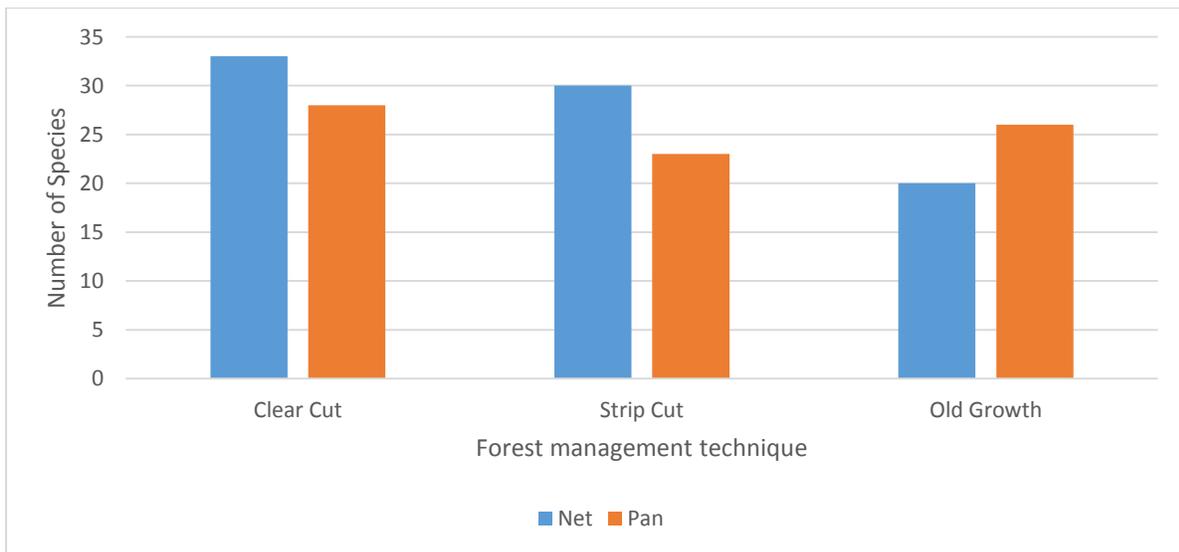


Figure 7a. Species richness collected in each management technique. The blue bars indicate that the pollinator was collected via the netting method whereas the orange bars indicate the pollinator was collected via the pan trap method.

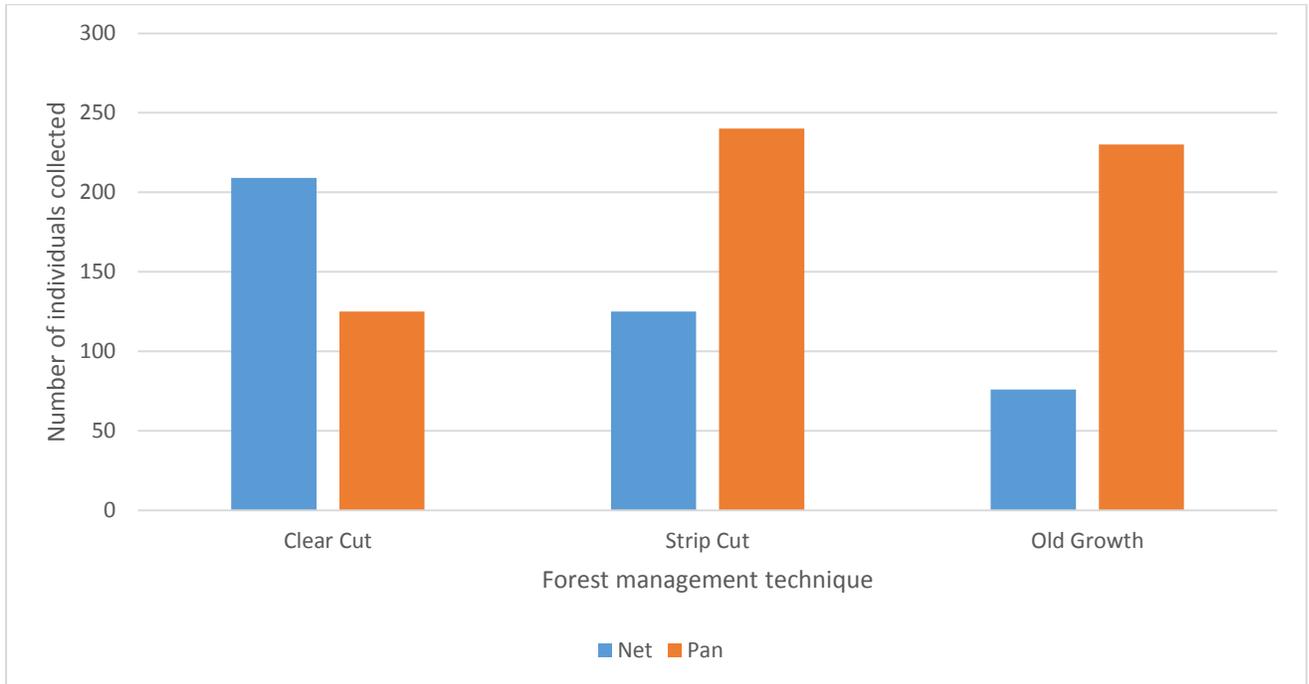


Figure 7b. Pollinator abundance collected in each management technique. The blue bars indicate that the pollinator was collected via the netting method whereas the orange bars indicate the pollinator was collected via the pan trap method.

When comparing collection methods, there was not one method that obviously collected more individuals or number of species. In the strip cut and clear cut sites, the netting method lead to a greater number of species collected whereas the the pan trap method collected more species in the old growth sites (Figure 7a). When comparing the number of individuals collected, the netting method lead to more individuals collected whereas in the strip cut sites and old growth sites, the pan traps collected more individuals (Figure 7b). The differences in number of individuals collected between methods was much greater than the differences in number of species.

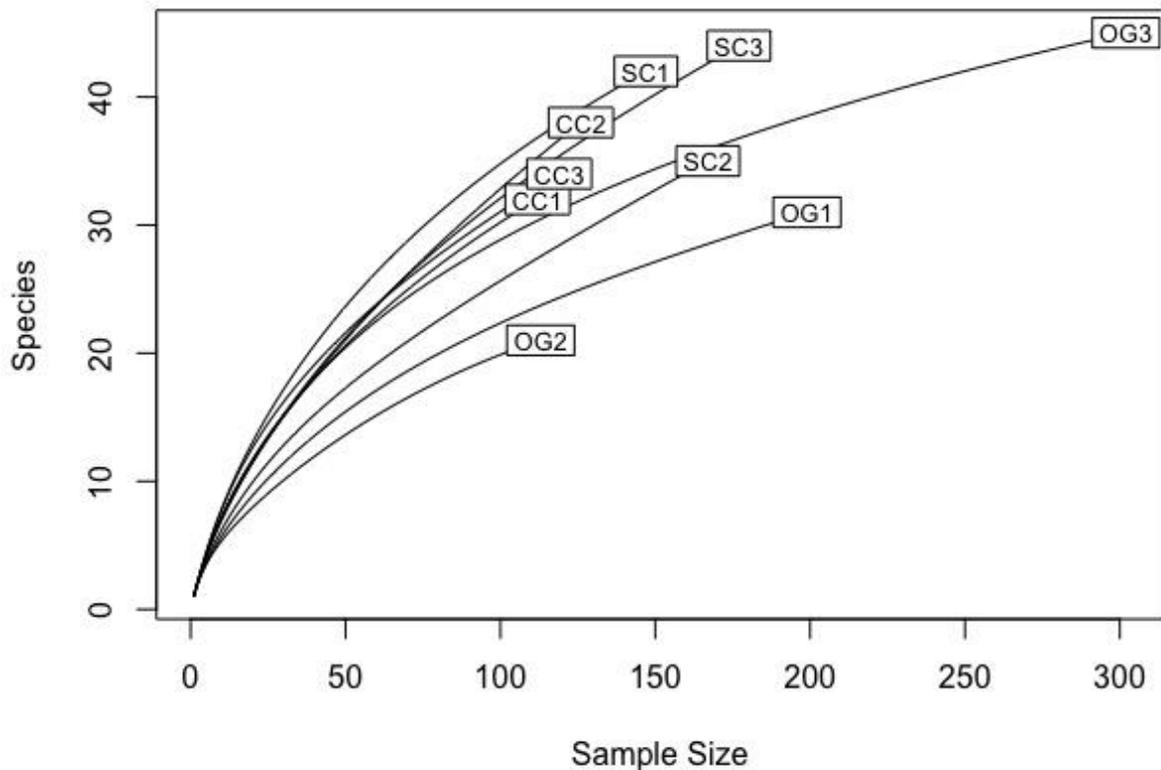


Figure 8. Rarefaction curve for each site where insects were collected. All species of insects identified were considered in this curve. The “CC” represents clear cut sites, the “SC” represents strip cut sites and the “OG” represents old growth sites.

Of all sites, the third old growth site was best sampled as the rarefaction curve is nearing an asymptotic height, which would mean that even if we had sampled more, the number of species collected would not increase dramatically. The first and second old growth site are also nearing a maximum height on the curve although their maximum number of species collected would be lower than all the other sites. All the strip cut sites and clear cut sites have not reached an asymptotic height, indicating that these sites were not adequately sampled (Figure 8).

Of the 126 species collected for all the insects identified, 15 species were only collected twice whereas 53 species were only collected once. The asymptotic species richness was estimated to be 219 species (Chao 1984). This difference in estimated species richness and collected species richness further indicates that this property was potentially under sampled.

Discussion

The objective of this research was to determine if forestry impacted pollinator diversity and pollinator community composition. Secondary objectives include looking at species specific phenology as well as comparing collection methods. Overall, only one of the four diversity indices was found to have significant differences in two of the three management techniques, thus concluding that there is likely not a large difference in number of species and in abundance collected between strip cuts, clear cuts and old growth forestry practices (Figure 4).

However, the community composition between the management techniques are significantly different, indicating that although they have similar diversity indices, the pollinating species are generally different in the three habitats following forestry management (Figure 5). This conclusion highlights the importance of habitat heterogeneity in an area as different habitats open up the potential for more niches to be filled and thus can support different types of pollinators (Rubene et al. 2015). This phenomenon can be seen when comparing the species across the three types of forest management types. The pollinators in the clear cut sites were mostly large bee species (*Bombus sp.*, *Apis mellifera*, *Megachile sp.*) whereas the strip cut generally had medium sized bee species (*Lasioglossium sp.*, *Halictus sp.*, *Andrena sp.*) and the old growth generally had the most beetles (*Trichontinus assimilis*), truebugs (*Neokolla hieroglyphica*) and small green bees (*Augochlora pura*, *Augochlorella striata*) (Figure 4.).

The differences in community structure may be from the amount of available sunlight as well as the abundance of fallen wood in a given site. The clear cut sites are fully exposed to sunlight which allows the seed bank in the soil to flourish. Additionally, since the clear cut sites do not have a previously established vegetation, weedy flowering species can grow in large colonies (Hengstum et al. 2013). These areas may be preferable to pollinators since there is often a greater abundance of pollen. Contrasting these areas are old growth forests where the canopy almost completely shades the forest floor which only permits shade tolerant flowers to grow (Devoto et al. 2012). However, decaying wood is more common in old growth forests since much of the wood is removed from a site following forestry. These decaying logs provide nesting habitat for many true bug, beetle and solitary bee species. The Hemiptera and Coleopterans also tend to be less capable of travelling vast distances, thus having a nesting habitat nearby may be vital to their survival and reproduction (Basilio et al. 2006). Due to their need for both flowering plants and woody habitat, an area that has been strip cut would ideally have the greatest abundance and species richness as seen in previous studies where removing patches of trees in a continuous landscape has increased bee abundance (Drever et al. 2006, Romey et al. 2007). Previous studies suggesting that forestry actually increases pollinator diversity have only looked at bee species (Devoto et al. 2012, Drever et al. 2006, Korpela et al. 2015, Romey et al. 2007). This research generally did not find a significant difference in diversity as forestry increases which may be because this study also collected pollinating true bugs and beetles which are more dependent on fallen wood.

In accordance with other studies, the old growth sites generally had few pollination interactions due to the number of wild flowers. The sites that were recently logged had many infrequent interactions and a few common interactions (Devoto et al. 2012, Memmott 1999). In all three

sites, the flowers pollinated by the most insect species were generally non-native plants (Figure 6). These non-native plant species may be acting as a reservoir for native pollinators, as they contain pollen sources to rely on when native wild flowers are not in bloom (Winfree et al. 2007). However, these non-native species may out compete native wild flowers since they can grow in large matted colonizes and eventually shading sunlight from the slower growing wild flowers. It is thus difficult to conclude if the introduction of non-native flowers in this area is beneficial or detrimental to pollinating communities, as they may temporarily provide a pollen source but also prevent native flower species from eventually establishing (Thompson et al. 2014).

When conducting a study across an entire season, it is often useful to observe at how pollinator populations change through the sampling season. Different groups of pollinators portrayed different phenological trends which may be interesting to investigate in future studies (Basillo et al. 2006). For example, two bumble bee species, *Bombus ternarius* and *Bombus vagans* increased in abundance each week and they were most abundant in the last week of sampling (Figure 4c). Although this study did not investigate why this occurred, future studies may include observing flower blooming fluctuations and when insect nesting occurs.

This study used two different methods to collect pollinators, a passive pan trap method and an active netting method. Although both methods were successful at collecting pollinators, they both have advantages and disadvantages. The netting method was more specific in only collecting pollinators and was also more successful at catching more species and more individuals in the clear cut sites. However, this method collected fewer individuals in the strip cut sites and old growth sites. The pan trap methods collected more pollinators in the old growth and strip cut sites, however, this method is much less specific and also collected many non-pollinating species. In accordance with similar studies, this study suggests that using both methods are useful, but perhaps additional effort should be put into the netting method which would maximize the number of pollinators collected (Moreira 1996, Potts et al. 2010). The use of a rarefaction curve is also essential in studies of diversity as it permits the researcher to know if adequate samples have been collected (Buddle et. 2005). In most sites additional sampling would have been necessary to better understand the pollinator communities.

This study lead to new insights on the population dynamics of less traditionally studied pollinators such as beetles and true bugs. Although these pollinators are not as efficient as wild bees, they still provide important services in plant reproduction of forested habitats (Faegri & Van Der Pijl 2013). The main outcome of this study suggests that although the different forest management techniques did not alter pollinator diversity, forestry did impact the types of pollinators in each habitat. Therefore, in order to maximize all species of pollinators, open habitats which provide productive wild flowers should be maintained but old growth forest habitats which offer many resources for pollinators should also be conserved (Korpela et al. 2014).

Appendix

Site	Latitude	Longitude	Elevation
Clear Cut 1	N45.755°	W74.861°	220m
Clear Cut 2	N45.754°	W74.86°	220m
Clear Cut 3	N45.752°	W74.863°	220m
Old Growth 1	N45.718°	W74.844°	200m
Old Growth 2	N45.755°	W74.823°	200m
Old Growth 3	N45.776°	W74.806°	200m
Strip Cut 1	N45.733°	W74.87°	195m
Strip Cut 2	N45.729°	W74.869°	195m
Strip Cut 3	N45.81°	W74.845°	220m

Appendix 1. Geographic location and elevation of the nine sites sampled.

	Clear Cut					Strip Cut					Old Growth				
	Site 1	Site 2	Site 3	Total	Mean	Site 1	Site 2	Site 3	Total	Mean	Site 1	Site 2	Site 3	Total	Mean
Total Species Richness	35	42	39	66	38	52	44	58	74	51.3	37	33	49	72	39.6
Pollinator Species Richness	32	39	34	61	35	47	37	48	52	44	31	20	41	40	30.6
Total Abundance	112	126	119	407	119	147	167	177	491	163.6	197	114	301	612	204
Pollinator Abundance	107	118	100	325	108.3	133	175	121	429	143	104	66	169	339	113
Shannon Diversity Index	2.840	2.941	3.10	N/A	2.961	3.297	2.608	3.379	N/A	3.095	2.795	2.038	2.228	N/A	2.354
Simpson Diversity Index	0.915	0.912	0.945	N/A	0.924	0.957	0.842	0.947	N/A	0.915	0.904	0.836	0.941	N/A	0.8936
Inverse Berger-Parker Index	4.956	4.740	7.857	N/A	5.851	7.315	2.765	5.360	N/A	5.147	4.065	3.090	6.750	N/a	4.635

Appendix 2. Species abundance, species richness and diversity indices for each management technique and site.

Literature Cited

- Aizen MA, Harder LD. 2009. The global stock of domesticated honey bees is growing slower than agricultural demand for pollination. *Current biology*. 19(11):915-918.
- Albano S, Salvado E, Borges PA, Mexia A. 2009. Floral visitors, their frequency, activity rate and Index of Visitation Rate in the strawberry fields of Ribatejo, Portugal: selection of potential pollinators. Part 1. *Advances in Horticultural Science*.238-245.
- Allsopp MH, De Lange WJ, Veldtman R. 2008. Valuing insect pollination services with cost of replacement. *PLoS One*. 3(9):e3128.
- Basilio AM, Medan D, Torretta JP, Bartoloni NJ. 2006. A year-long plant-pollinator network. *Austral Ecology*. 31(8):975-983.
- Beck J, Schwanghart W. 2010. Comparing measures of species diversity from incomplete inventories: an update. *Methods in Ecology and Evolution*. 1(1):38-44.
- Biesmeijer JC, Roberts S, Reemer M, Ohlemüller R, Edwards M, Peeters T, Schaffers A, Potts S, Kleukers R, Thomas C. 2006. Parallel declines in pollinators and insect-pollinated plants in Britain and the Netherlands. *Science*. 313(5785):351-354.
- Buddle CM, Beguin J, Bolduc E, Mercado A, Sackett TE, Selby RD, Varady-Szabo H, Zeran RM. 2005. The importance and use of taxon sampling curves for comparative biodiversity research with forest arthropod assemblages. *The Canadian Entomologist*. 137(1):120–127.
- Burkle LA, Marlin JC, Knight TM. 2013. Plant-pollinator interactions over 120 years: loss of species, co-occurrence, and function. *Science*. 339(6127):1611-1615.
- Cariveau DP, Williams NM, Benjamin FE, Winfree R. 2013. Response diversity to land use occurs but does not consistently stabilize ecosystem services provided by native pollinators. *Ecology letters*. 16(7):903-911.
- Chao A .1984. Nonparametric estimation of the number of classes in a population. *Scand J Stat* 11:265–270.
- Devoto, M., Bailey S, Craze P., & Memmott J. 2012. Understanding and Planning Ecological Restoration of Plant-pollinator Networks. *Ecology Letters* 15(4): 319-28.
- Dormann, C.F., Fruend, J., Bluethgen, N. & Gruber B. 2009. Indices, graphs and null models: analyzing bipartite ecological networks. *The Open Ecology Journal*, 2: 7-24.

Doyon F, Gagnon D, Giroux J-F. 2005. Effects of strip and single-tree selection cutting on birds and their habitat in a southwestern Quebec northern hardwood forest. *Forest Ecology and Management*. 209(1–2):101-116.

Dicks LV, Abrahams A, Atkinson J, Biesmeijer J, Bourn N, Brown C, Brown MJF, Carvell C, Connolly C, Cresswell JE, et al. 2013. Identifying key knowledge needs for evidence-based conservation of wild insect pollinators: a collaborative cross-sectoral exercise. *Insect Conservation and Diversity*. 6(3):435-446.

Drever CR, Peterson G, Messier C, Bergeron Y, Flannigan M. 2006. Can forest management based on natural disturbances maintain ecological resilience? *Canadian Journal of Forest Research*. 36(9):2285-2299.

Faegri K, Van Der Pijl L. 2013. *Principles of pollination ecology*. Elsevier.

Fishbein M, Venable DL. 1996. Diversity and temporal change in the effective pollinators of *Asclepias tuberosa*. *Ecology*. 77(4):1061-1073.

Garibaldi LA, Carvalheiro LG, Leonhardt SD, Aizen MA, Blaauw BR, Isaacs R, Kuhlmann M, Kleijn D, Klein AM, Kremen C. 2014. From research to action: enhancing crop yield through wild pollinators. *Frontiers in Ecology and the Environment*. 12(8):439-447.

Gleason HA, Cronquist A. 1963. *Manual of vascular plants of Northeastern United States and adjacent Canada*. Princeton, N.J.: Van Nostrand.

Gilgert W, Vaughan M. 2011. The Value of Pollinators and Pollinator Habitat to Rangelands: Connections Among Pollinators, Insects, Plant Communities, Fish, and Wildlife. *Rangelands*. 33(3):14-19.

Graham-Sauvè L, Work TT, Kneeshaw D, Messier C. 2013. Shelterwood and multicohort management have similar initial effects on ground beetle assemblages in boreal forests. *FORECO Forest Ecology and Management*. 306:266-274.

Hengstum T, Hooftman DAP, Oostermeijer JGB, Tienderen PH, Mack R. 2014. Impact of plant invasions on local arthropod communities: a meta-analysis. *JEC Journal of Ecology*. 102(1):4-11.

Ishida C, Kono M, Sakai S. 2009. A new pollination system: brood-site pollination by flower bugs in *Macaranga* (Euphorbiaceae). *Annals of botany*. 103(1):39-44.

Jauker F, Bondarenko B, Becker HC, Steffan-Dewenter I. 2012. Pollination efficiency of wild bees and hoverflies provided to oilseed rape. *Agricultural and Forest Entomology*. 14(1):81-87.

Jost, L. (2006). Entropy and diversity. *Oikos*, 113(2), 363-375.

Kluser S, Peduzzi P. 2007. Global pollinator decline: a literature review.

Korpela, E.-L., Hyvönen, T., & Kuussaari, M. (2015). Logging in boreal field-forest ecotones promotes flower-visiting insect diversity and modifies insect community composition. *Insect Conservation and Diversity*, 8(2), 152-162.

Lencinas MaV, Martínez Pastur G, Gallo E, Cellini JM. 2014. Decreasing negative impacts of harvesting over insect communities using variable retention in southern Patagonian forests. *J Insect Conserv Journal of Insect Conservation : An international journal devoted to the conservation of insects and related invertebrates*. 18(3):479-495.

Lindenmayer DB, Margules CR, Botkin DB. 2000. Indicators of Biodiversity for Ecologically Sustainable Forest Management. *Conservation Biology*. 14(4):941-950.

Lowenstein DM, Matteson KC, Minor ES. 2015. Diversity of wild bees supports pollination services in an urbanized landscape. *Oecologia*. 179(3):811-821.

Mace GM, Norris K, Fitter AH. 2012. Biodiversity and ecosystem services: a multilayered relationship. *Trends in ecology & evolution*. 27(1):19-26.

Maleque MA, Kaoru M, Hiroaki TI, Maeto K, t, Ishii Hiroaki T. 2009. Arthropods as bioindicators of sustainable forest management, with a focus on plantation forests. *Applied entomology and zoology*. 44(1):1-11.

Memmott J. 1999. The structure of a plant-pollinator food web. *Ecology letters*. 2(5):276-280.

Memmott, J., Craze, P. G., Waser, N. M., & Price, M. V. (2007). Global warming and the disruption of plant–pollinator interactions. *Ecology Letters*, 10(8), 710-717.

Moreira EF, Santos RLdS, Penna UL, Angel-Coca C, de Oliveira FF, Viana BF. 2016. Are pan traps colors complementary to sample community of potential pollinator insects? *Journal of Insect Conservation*. 20(4):583-596.

Natural Resources Canada: Forests (Canada). 2016. The state of Canada's Forests, Annual Report for 2016 [Internet]. Natural Resources Canada; [cited 2017 March 6].

Oksanen J, Blanchet GF, Friendly M, Kindt R, Legendre P, McGlenn D, Minchin PR, O'Hara RB, Simpson GL, Solymos P, Stevens HH, Szoecs E and Wagner H. 2016. vegan: Community Ecology Package. R package version 2.4-1.

Ollerton J, Winfree R, Tarrant S. 2011. How many flowering plants are pollinated by animals? *Oikos*. 120(3):321-326.

Overall LM, and EJ Rebeck. 2015. Seasonal Abundance and Natural Inoculativity of Insect Vectors of *Xylella fastidiosa* in Oklahoma Tree Nurseries and Vineyards. *Journal of Economic Entomology*. 108 (6): 2536-45.

Popic TJ, Davila YC, Wardle GM. 2013. Evaluation of common methods for sampling invertebrate pollinator assemblages: net sampling out-perform pan traps. *PloS one*. 8(6).

Potts SG, Biesmeijer JC, Kremen C, Neumann P, Schweiger O, Kunin WE. 2010. Global pollinator declines: trends, impacts and drivers. *Trends in ecology & evolution*. 25(6):345-353.

Proctor MCF, Yeo P, Lack A. 1996. *The natural history of pollination*. Portland, Or.: Timber Press.

R Development Core Team. 2016. *R: A language and environment for statistical computing*. R Foundation for Statistical Computing, Vienna, Austria. ISBN 3-900051-07-0, URL <http://www.R-project.org>.

Romey WL, Ascher JS, Powell DA, Yanek M. 2007. Impacts of Logging on Midsummer Diversity of Native Bees (Apoidea) in a Northern Hardwood Forest. *Journal of the Kansas Entomological Society*. 80(4):327-338.

Rubene D, Schroeder M, Ranius T. 2015. Diversity patterns of wild bees and wasps in managed boreal forests: effects of spatial structure, local habitat and surrounding landscape. *Biological Conservation*. 184:201-208.

Rundlöf M, Andersson GK, Bommarco R, Fries I, Hederström V, Herbertsson L, Jonsson O, et al. 2015. Seed coating with a neonicotinoid insecticide negatively affects wild bees. *Nature*. 521: 77-80.

Thompson ID, Okabe K, Parrotta JA, Brockerhoff E, Jactel H, Forrester DI, Taki H. 2014. Biodiversity and ecosystem services: lessons from nature to improve management of planted forests for REDD-plus. *Biodiversity and conservation*. 23(10):2613-2635.

Thien LB, Azuma H, Kawano S. 2000. New Perspectives on the Pollination Biology of Basal Angiosperms. *International Journal of Plant Sciences*. 161(S6):S225-S235.

Tylianakis JM. 2013. The global plight of pollinators. *Science*. 339(6127):1532-1533.

Ugland KI, Gray JS, Ellingsen KE. 2003. The species-accumulation curve and estimation of species richness. *Journal of Animal Ecology*. 72(5):888-897.

Winfrey, R., Griswold, T., & Kremen, C. (2007). Effect of Human Disturbance on Bee Communities in a Forested Ecosystem

Wheeler AG. 2001. Biology of the plant bugs (Hemiptera: Miridae): pests, predators, opportunists. Cornell University Press.