

**NATIVE POLLINATORS IN MANAGED ECOSYSTEM FOREST  
FRAGMENTS**

by

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A thesis submitted to the Faculty of the University of Delaware in partial fulfillment of the requirements for the degree of Honors Bachelor of Sciences in Wildlife Conservation with Distinction.

Spring 2012

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## **ACKNOWLEDGMENTS**

I cannot thank Dr. Deborah Delaney enough for being an amazing advisor, mentor, and friend throughout this research project and who has instilled in me an incredible and lifetime love of bees.

My experience through the FRAME project would not have been the same without Dr. Greg Shriver and Dr. Vince D'Amico, who both organized the entire research effort and offered me comic relief in times of need. As well as Christine Rega, who received her Master's degree, for painstakingly organizing the data collections for the FRAME project and not hesitating to offer me words of wisdom.

Sam Droege from the USGS Native Bee Inventory and Monitoring lab could not have been more accommodating when Dr. Delaney and I met with him. His collection protocol manual allowed the formation of sound methodology and his dedication to native bees made me cherish how valuable this research is.

I am grateful to the University of Delaware Undergraduate Research Program and the FRAME grant for funding my research during the summer of 2011. Without said funding this project could not have been made possible.

I would like to extend my thanks to my senior thesis committee – Dr. Delaney, Dr. Shriver, and Dr. Kniel for offering me their support and taking time out of their busy schedules to help me complete a comprehensive project that encompasses so many of the skills I have learned at the University of Delaware.

Lastly, I am sorry to all the bees whose lives I took during this process. Please know that it was to only secure your progeny a better future.

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## ABSTRACT

Due to the increasing ecological importance of native pollinators as honey bee populations continue to decline, this study sought to inventory and compare native bee abundance, richness, and diversity in 13 northeastern Delaware forest fragments. The main objectives of this research were to understand relationships between native bees and the plant community and apply any findings to better manage habitat fragments in the area. Native bees were collected from March to September 2011 using bowl traps and netting techniques. Specimens collected were identified to family and Apidae specimens were identified to genus. Vegetation data and ArcGIS 100m buffers were used to characterize fragments within, and landuse surrounding fragments, specifically assessing the percent human development, impervious surface, and agriculture. Results revealed a temporal significance between specimen numbers collected before and after May 1<sup>st</sup>. Andrenidae represented close to half of the composition of total bee families, and the cleptoparasitic genus *Nomada* dominated Apidae composition at 88 and 80.21%. The regression results showed weak trends suggesting that native bees may utilize human disturbed habitats and tolerate certain levels of disturbance more than previously thought. Possible reasons for this tolerance could be the supply of consistent forage and nesting sites within and around fragments. The study helps build baseline data for native bee families in future urban fragment research.

## Chapter 1

### INTRODUCTION

Native bees refer to any wild bees (not including the European honey bee; *Apis mellifera* L.) that pollinate wild and cultivated plants. There are approximately 4,000 species of North American native bees and around 200 recorded in Delaware (Sarver 2007). Greater than two-thirds of North American native bees are ground nesters while the remaining third nest in the pith of stems and twigs, pre-existing cavities, rotten logs, and grass tussocks (Cane 1991). Most of the native bees are solitary and nest aggregately in favored sites rather than forming large colonies. Some common native bee families include Apidae (including genera *Xylocopa*, *Bombus*, *Ceratina*, and *Nomada*), Andrenidae, Colletidae, Halictidae, and Megachilidae.

Various groups of native bees including, but not limited to, *Bombus* or bumble bees, mason bees, squash bees, sweat bees, and long-horned bees are vital for pollination of over 100 crops. In 2000 in the United States native bees pollinated an estimated \$3 billion of crops (Sarver 2007). Honey bee decline continues due to mite overloads and disease (Williams 2010) and as native bees can increase crop yields, more research needs to be focused on the ecology and sustainability of native bee populations and community structure. In a watermelon crop pollination study, Winfree et al. (2007a) concluded that native bees could provide insurance against honey bee loss due to two findings that native bees thrived in human disturbed habitats associated with canopy closure in native woodlands which provides few flowers for foraging (Kremen et al. 2007) and that a high dispersion of natural habitat fragments may



support high biodiversity of bees even if overall natural habitat composition is low (2007a).

Agriculture and urbanization pose one of the greatest risks to natural ecosystems through habitat fragmentation (Samways 1994). Worldwide, more than half of the temperate broadleaf and mixed forest habitats have been removed or fragmented (Wade et al 2003). Remnant patches of habitat within a disturbed environment are essentially islands of refuge for various wildlife species. Additionally, intensified agriculture harms native pollinators through displacement of forage, mortality or altered forage behavior due to pesticide use, and the destruction of nesting sites through tillage, removal of woody vegetation, irrigation, and decreased diversity of habitat (Kremen et al. 2007).

Native flora and fauna within fragmented landscapes experience abundance, distribution, ecosystem function, and community composition alterations. Modifying fragment shapes and sizes increases edge habitat which in turn creates ideal habitat conditions for exotic and invasive species (Faulkner 2004). Extinction rates are positively associated with the degree of fragmentation; as fragmentation increases, so does the extinction rate.

Insects' responses to disturbance vary considerably depending upon the type, degree, and extent. Depending on the tolerance of the species to change, some species may locate towards the disturbance and thrive whereas others may die off. Some insects have demonstrated some resilience to fragmentation in their ability to migrate between patches, although roads have proven to be effective barriers to most insect movement (Samways 129). The effect on the community includes impacts to parasitoids, predators, and plant food sources (Samways 140).

Native bees have been largely overlooked in research studying the impacts of habitat fragmentation until recently. James Cane believes that conservationists have been too quick to consider fragmentation a “broadly deleterious” condition to all organisms when there is evidence that some species actually thrive in such habitat (Cane 2001). For example, native bees may actually prefer fragmented habitat if it provides their basic requirements such as a diversity of flowering plants for forage and ample nesting sites. As Cane (2001) points out, most native bee studies have defined fragmentation for bees as patches of flowering plants without addressing nesting site availability which may be the ultimate limiting factor for bee species rather than forage patches, due to smaller bees’ remarkable forage distances of an average 2.8 km.

Kremen et al. (2007) synthesis of multiple studies summarizes possible explanations of differing results with native bee response to disturbance. Pollinator richness may increase initially in response to environmental disturbance that is intermediate and/or repetitious (Steffan-Dewenter 2002). However, as the degree of disturbance increases, or as the habitat reaches climax status with moderately few plant species (Winfree et al. 2007b), the pollinator richness begins to decrease. “Bee species richness may be maximized at an intermediate level of human disturbance, with negative effects only occurring when natural land cover falls below some threshold” (Winfree et al. 2007b). The system in which the study was completed had 25-99% forest cover which may limit flowering plants throughout the summer; however, the researchers state that despite the peak bloom in the spring, forests still had lower bee abundance than the other habitat tested including agriculture, and low and high density housing in the Pinelands of New Jersey.

In a meta-analysis on bees and fragmentation, it was found that anthropogenic disturbance had a significant negative effect on native bee abundance whereas agriculture, logging, and fire did not (Winfree et al. 2009). Though in all disturbance types *Bombus* species were significantly negatively affected. Most interesting, is that though there was not a significant difference between social and solitary bees' abundance; social bees were significantly negatively affected by disturbance whereas solitary bees were not. Anthropogenic disturbance was also the only disturbance type that significantly negatively affected bee species richness. The richness of social bees was significantly negatively affected by disturbance whereas solitary bees were not; however, there was not a significant difference between the two groups. In extreme habitat (defined as an area less than or equal to 1 hectare, surrounded by less than or equal to 5% native cover, or a site greater than or equal to 1 km from natural habitat) both bee abundance and richness were significantly negatively impacted whereas in moderate habitat (not defined) they were not.

*Bombus* species are the most effective pollinators of plants on a per-visit basis (Kremen et al. 2002). However, *Bombus* species are also the most negatively affected by loss of native vegetation (Winfree and Kremen 2008). *Bombus*, or bumblebees, like honeybees, are eusocial meaning that there is division of labor within a colony, overlapping of generations, and cooperative brood care. *Bombus* species are native to North America and form annual colonies whereas *Apis* is introduced, invasive and forms perennial colonies.

Foraging distance information is important for understanding native bees' response to habitat fragmentation and plant community restructuring (Greenleaf et al. 2007). Greenleaf et al. (2007) found that as native bee body size increased, so did the

foraging distance. In a German study on honeybee foraging distance, it was concluded that the mean foraging distance of all bees did not significantly differ from that of nectar collecting bees between simple (agricultural areas and large patch size) and complex landscapes (smaller patch sizes, high habitat diversity, and large amounts of semi-natural habitat) though they did differ temporally (Steffan-Dewenter and Kuhn 2003). Pollen-collecting bees had significantly increased foraging distances in simple, rather than complex landscapes. These bees also had the highest foraging distances recorded during the month of June when floral resources were scarce since honeybees store more nectar than pollen.

Some 42% of the world's threatened and endangered plant and animal species are considered to be at risk primarily due to competition with exotics (Wilcove et al. 1998). *Rosa multiflora* (commonly referred to as multiflora rose) was introduced into the United States from Japan for use as an ornamental rose root stock in 1866 (USDA 2005). In the 1930s the US Soil Conservation Service applauded the planting of multiflora rose to help control erosion. However, the plant's ability to vigorously grow in disturbed and open habitat, and produce some 500,000 to 1 million seeds a year, which are viable in the soil for up to 20 years, has lead to it being classified as a noxious invasive. There is currently 45 million acres of multiflora rose in the eastern half of the United States (Loux et al. 2005). Multiflora rose is notorious for forming dense, thorny thickets in which it replaces native vegetation, inhibiting tree growth by competing for resources such as sunlight, water and nutrients, and altering wildlife behavior (USDA 2005).

Multiflora rose flowers bloom in May and June (USDA 2005). Lee et al. (1995) found that multiflora rose in Korea, part of its native habitat, was primarily

pollinated by *Apis mellifera*. Jesse et al. (2006) studied the multiflora rose flower visitation rates by invertebrates using sticky cards in Iowa in 2002 and 2003. The most common orders observed overall were Coleoptera, Diptera, Hemiptera, Hymenoptera, Mecoptera, and Thysanoptera. In 2002, the most common insect visitors were Syrphids and other flies. While *Apis mellifera* and *Bombus* spp. were observed, they were not common. In 2003, Syrphid flies comprised 43% of the insects that visited multiflora rose flowers, 35% were other Dipteran species, and 21% included *Apis mellifera*, *Bombus*, and solitary bees. The researchers were not surprised to observe the introduced *Apis mellifera*, a generalist, pollinating an introduced plant species. Thus, the use of generalist pollinators by multiflora rose has seemingly allowed it to sidestep pollination as its limiting factor (Jesse et al. 2006). However, the effects of multiflora rose on native bee populations in forest fragments are still largely unknown.

Of the 4-5,000 bee species in North America, fifteen percent are parasitic (Bohart 1970). Cleptoparasitic bees lay their eggs in the nest of an unsuspecting host and their larvae will eventually kill the host larvae and consume the provided food (Bohart 1970). The most ancient and diverse group of parasitic bees is the subfamily Nomadinae. *Nomada*, the largest parasitic bee genus (with over 100 North American species), includes hosts from six of the major bee families including many members of the large *Andrena* genus (family Andrenidae). Halictidae, Melittidae, and Anthophoridae are also popular host families (Bohart 1970).

In a tropical host-parasitoid food web study, Tylianakis et al. (2007) found that in modified habitats there were higher ratios of parasitoid to host species and increased parasitism rates. As habitat modification increased so did the vulnerability of many of the host bee species. Parasitoid specialization on a particular species lessened the

parasitism pressures on alternative hosts (Tylianakis et al. 2007). Such food web alterations could have devastating impacts on ecosystems where the host bees provide vital pollination and ecosystem services.

The purpose of this study was to (1) inventory and compare the native bee species abundance, richness, and diversity in 13 forest fragments in northeastern Delaware to assess their status, (2) help to understand ecological interactions between native bees and their environment such as the soil and surrounding plant community, and (3) apply the findings to better manage woodlots and habitat patches experiencing an increase in urban development and invasive species encroachment.

## **Chapter 2**

### **METHODOLOGY**

#### **Collection Protocol**

This study was part of the larger FRAME project (Forest fragments in Managed Ecosystems) taking place at the University of Delaware. The FRAME project includes data collections in 21 established forest fragments throughout Newark, DE. Of the 21 fragments, this study included 13 sites and then narrowed the focus to 5 sites monitored on a more consistent basis. Each site was gridded at 25 m intervals using ArcGIS software, except the University of Delaware's Ecology Woods which was gridded previously at 50 m intervals. Ten active sites were selected at random for each site and were buffered on all sides by inactive points. The native pollinator collections used the 2010 active points for which vegetation and soil data had already been collected.

Soil data such as pH were measured by taking soil samples at each active point. Vegetation parameters used along with this study were measured in terms of ground cover surveys. At each active point a 2.5 m radius was determined and the percentage of ground covered by vegetation was determined. Then, within that same radius, the shrubs above 1 m tall and less than 5 m tall were identified. The numbers of stems produced by each plant greater than 1 m were counted and for larger shrubs the

estimated numbers of stems produced in a given area were multiplied by the area of the shrub.

ArcGIS software allowed for the determination of parameters such as the percentage of human development, agricultural land, and impervious surface within a 100 m buffer surrounding the whole site. 100 m buffers were chosen because anything larger would have caused overlapping of site buffers in some locations.

Native pollinator sampling began at the end of March 2011 and continued throughout the summer till September 2011 at two week intervals. Bee bowls were used to sample the smaller bees in the fragments by attracting bees to the colored bowls and killing and preserving them upon impact. Bee bowls were created by filling 1.5 ounce solo cups with an anti-freeze, soap, and bleach mixture which kills the bees. The soap used was Blue Dawn original formula which decreases the surface tension allowing even the smallest of the trapped insects to sink below the surface. The propylene glycol (red standard automobile anti-freeze) was used to trap and kill the bees and a splash of bleach was used to simply rid the mixture of the red color. This method is an approved pollinator collection protocol by Sam Droege at the USGS Native Bee Inventory and Monitoring lab (Droege 2012). Three bowls were set on the ground at each point within each site – bright blue, bright yellow, and plain white (the wavelengths that bees can perceive). The bowls were set 3 meters apart so as not to interfere with the bees collected by each.

Larger bees were sampled by time constrained netting at each site every two weeks, following the same schedule as the bee bowl collections. The netting was



completed for 30 minutes at each site on blooming vegetation where bees were visible and the plant species were recorded.

During each bee bowl collection, the Hymenoptera were collected in jars filled with alcohol until they could be sufficiently washed, blow-dried, and pinned according to the accepted protocol (Droege 2012). The Hymenoptera that were collected during netting were placed in kill jars charged with ethyl acetate and later pinned without washing. The specimens were then identified to family and some to genera where applicable.

### **Study Area**

Thirteen forest fragments ranging in size from 4.5 ha to 855.9 ha in Newark, Delaware were sampled. Figure 1 shows the distribution of the FRAME sites used in this study. Each of the fragments contained a running stream and each fragment contained vegetation common to both the Piedmont and Coastal Plain region due to Newark's unique geographic location. The degree of total ground cover varied throughout the study areas, ranging from 40.89 - 0.67% while non-native ground cover ranged from 13.85 - 0%. Multi-flora rose ground cover ranged from 10.16 - 0%. Surrounding impervious surface ranged from 47.53 – 0%, surrounding agriculture ranged from 32.04 – 0%, and surrounding human development ranged from 93.51 – 0%. These statistics can be referenced in Table 1.

Ecology Woods, which is part of the University of Delaware Experimental Farm, represents the historic landscape transformation of Newark. Most of the patch consists of 130-year old second growth, mixed deciduous forest while some parts contain 70-year old second growth forest (Gorman and Roth 1989). The over story was dominated by tulip poplar (*Liriodendron tulipifera*), sweet gum (*Liquidambar styraciflua*), oaks (*Quercus* spp.), hickories (*Carya* spp.), American beech (*Fagus grandifolia*), and red maple (*Acer rubrum*). The mid-canopy contained flowering dogwood (*Cornus florida*), blackgum (*Nyssa sylvatica*), and blue beech (*Carpinus caroliniana*). Common shrubs were sweet pepperbush (*Clethra alnifolia*), spicebush (*Lindera benzoin*), arrowwood (*Viburnum dentata*), multiflora rose (*Rosa multiflora*), and greenbrier (*Smilax* spp.). Additionally, there was seasonal skunk cabbage (*Symplocarpus foetidus*), and poison ivy (*Toxicodendron radicans*) (Roth and Johnson 1993). The woodlot was fenced in and left undisturbed since 1979 and now it is ninety-five percent surrounded by agricultural and athletic fields (Roth and Johnson 1993).

The other 12 sites used in this study follow a similar life history with varying degrees of surrounding development, disturbance, ground cover, and non-native species encroachment.

### **Statistical Analysis**

Due to difficulties with sampling consistencies for all 13 sites, only 5 of the sites were consistently sampled at 2-week intervals throughout the entire collection

period. Bee data from all 13 sites was adjusted for visitation effort by the date the bowl was emptied of specimens and recharged. Thus, the number of bees collected at each site was divided by the number of bowls that were sampled during that site visit.

Due to a distinct temporal segregation between bee collection numbers before and after May 1<sup>st</sup>, the data was analyzed as two separate entities to obtain more descriptive results. Linear regressions for total bees from March to May and from May to September were run against H<sup>+</sup> ions (pH converted to a linear schedule), patch size, percent human development, agriculture, and impervious surface within a 100 m buffer of the site, percent total ground cover, non-native ground cover, multi-flora rose ground cover, and non-native stems. SPSS and JMP Pro 9 statistical software were used for this data analysis. Additionally, individual bee families and genera were run against each of the listed variables as well as *Nomada* against the other bee families as they are a cleptoparasitic bee and may affect other native bee numbers.

Members of the family Vespidae were accounted for in the collection data and run through linear regressions with the variables as they represent important and abundant Hymenopterans in forest fragment ecosystems.

In addition, a Shannon Diversity Index was calculated for the five sites that were most consistently sampled throughout the entire collection period. An index for total bee family diversity as well as Apidae genera diversity was calculated.

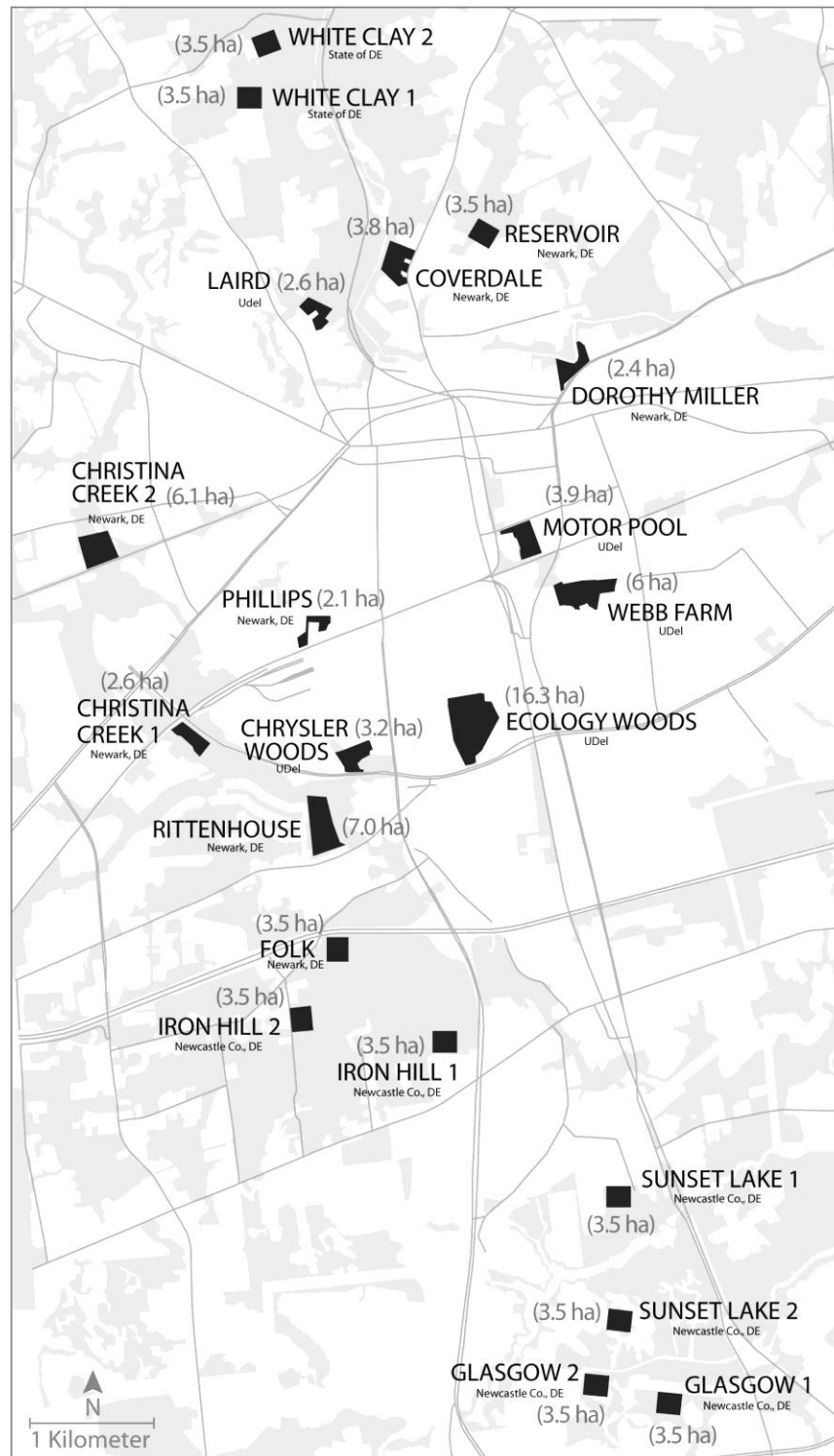


Figure 1 A map of the FRAME sites throughout Newark, DE. Of the 21 sites, 13 were used in this study.

Table 1 Surrounding development, vegetation composition, and study area size statistics for the 13 sites sampled

Site	Patch size (ha)	Surrounding Impervious Surface	Surrounding Agriculture	Surrounding Human Development	% Total Ground Cover	% Non-Native Ground Cover	% Multi-flora Rose Ground Cover
Dorothy Miller	68.13	24.54%	0.00%	55.79%	40.89	8.38	1.62
Ecology Woods	16.33	8.95%	32.04%	56.80%	30.69	8.24	3.53
Folk	855.9	18.83%	0.00%	42.34%	5.39	0.62	0.00
Glasgow 1	188.64	0.87%	0.84%	1.68%	9.67	0.00	0.00
Glasgow 2	188.64	1.24%	10.26%	11.11%	22.33	0.00	0.00
Iron Hill 1	855.9	2.67%	12.20%	17.07%	26.94	0.00	0.00
Iron Hill 2	855.9	0.91%	0.00%	13.68%	25.26	13.85	0.00
Motorpool	4.99	47.53%	0.00%	93.51%	24.45	10.86	4.32
Phillips	4.5	25.89%	0.00%	82.35%	21.75	13.16	10.16
Reservoir	49.23	12.12%	0.00%	40.34%	16.75	6.13	0.00
Rittenhouse	92.7	2.97%	0.00%	22.22%	11.64	1.46	0.38
Sunset Lake 2	188.64	0.00%	19.67%	0.00%	0.67	0.00	0.00
Webb Farm	9.63	15.94%	12.30%	52.41%	25.95	5.72	0.41

## Chapter 3

### RESULTS

#### General Trends

Overall adjusted total bee counts per site throughout the entire course of the collection period, from March to September 2011, ranged from around 0.5 bees in Iron Hill-1 to 6 bees in Dorothy Miller. Because clear significant differences were noted between the numbers of bees collected from March to May 1<sup>st</sup> and from May 1<sup>st</sup> to September, the two groups were separated throughout the data analysis.

Figure 7 illustrates the basic bee family breakdown of the collections before and after May 1<sup>st</sup>. Between the two time periods Andrenidae and Colletidae composition decreased from 64.67 – 43.86% and 2.56 – 0%, respectively. Andrenidae comprised the majority of total bees collected throughout the entire collection period. Halictidae and Apidae composition increased from 5.13 – 14.27% and 18.66 – 33.39%, respectively. Representatives from the family Megachilidae stayed fairly constant between both sampling periods.

Figure 8 depicts the breakdown of Apidae genera over the two time periods. Interestingly, *Xylocopa* comprised 0% of the bees collected in both groups. *Ceratina* composition increased from 9.60 – 18.39% while *Nomada* comprised the majority of the Apidae collected and remained fairly constant, decreasing from 88 – 80.21% between the two time periods.

### Shannon Diversity Index

Tables 2 and 3 contain the results of the Shannon Diversity Index for the 5 sites that were most consistently sampled for the entire collection period. The diversity index was computed regardless of the two defined temporal groups. Family richness and evenness are taken into account in the calculation of the diversity index. Evenness is calculated on a scale from 0 to 1, with 1 being completely even. For bee family diversity Iron Hill-2 had the highest diversity with 0.498 followed by a close second of Folk with 0.497. Rittenhouse, Ecology Woods, and Iron Hill-1 had respective diversities of 0.355, 0.318, and 0.217. For Apidae genera diversity (*Apis*, *Xylocopa*, *Bombus*, *Ceratina*, and *Nomada*) Ecology Woods, Folk, and Iron Hill-1 had diversities of 0 because there was only 1 genus present. Rittenhouse had a diversity of 0.09 and Iron Hill-2 had the highest diversity of 0.343.

### Regressions

Tables 4, 5, and 6 contain the regression results for before and after May 1<sup>st</sup> that were either significant or approaching significance. Surprisingly, H+ concentration and percentage of agriculture within a 100 m buffer of the site were never significant. Total bees vs. human development, though it approached significance before May 1<sup>st</sup>, changed between the dual group analyses. Before May 1<sup>st</sup> bee numbers approached a negative relationship with human development ( $p=0.0656$ ,  $r^2 = 0.0428$ ,  $\beta = -0.0107$ ); whereas, after May 1<sup>st</sup> bee numbers were positively associated with human development ( $p=0.0208$ ,  $r^2 = 0.0290$ ,  $\beta = 0.0272$ ). Apidae especially follows a similar trend vs. human development. Before May 1<sup>st</sup> Apidae negatively associates with human development ( $p = 0.0040$ ,  $r^2 = 0.1012$ ,  $\beta = -1.9011$ )

but after May 1<sup>st</sup> positively associates with human development ( $p=0.0451$ ,  $r^2 = 0.0219$ ,  $\beta = 0.6133$ ).

Total bees vs. patch size were significantly positively associated before May 1<sup>st</sup> ( $p=0.0282$ ,  $r^2 = 0.0602$ ,  $\beta = 1.5056$ ) though there was no significance found after May 1<sup>st</sup>. Apidae also had a positive association before May 1<sup>st</sup> ( $p=0.0229$ ,  $r^2 = 0.0646$ ,  $\beta = 0.0128$ ); however, after May 1<sup>st</sup> Apidae approached a significant, negative association with patch size ( $p = 0.0611$ ,  $r^2 = 0.0191$ ,  $\beta = -0.0003$ ).

Before May 1<sup>st</sup>, *Apis* and Colletidae were significantly positively associated with non-native ground cover ( $p = <0.0001$ ,  $r^2 = 0.1946$ ,  $\beta = 0.0012$  &  $p= <0.0001$ ,  $r^2 = 0.1810$ ,  $\beta = 0.0057$  respectively). Notably, these positive associations were relatively strong with 19.46% and 18.10% of the variation of the data explained by non-native ground cover.

Regressions were run for patch size against all of the vegetation parameters independent of the two temporal groups. Patch size vs. multi-flora rose ground cover and patch size vs. non-native stems resulted in negative associations ( $p=0.0003$ ,  $r^2 = 0.0700$ ,  $\beta = -0.0015$  &  $p= <0.0001$ ,  $r^2 = 0.1190$ ,  $\beta = -0.0173$  respectively).



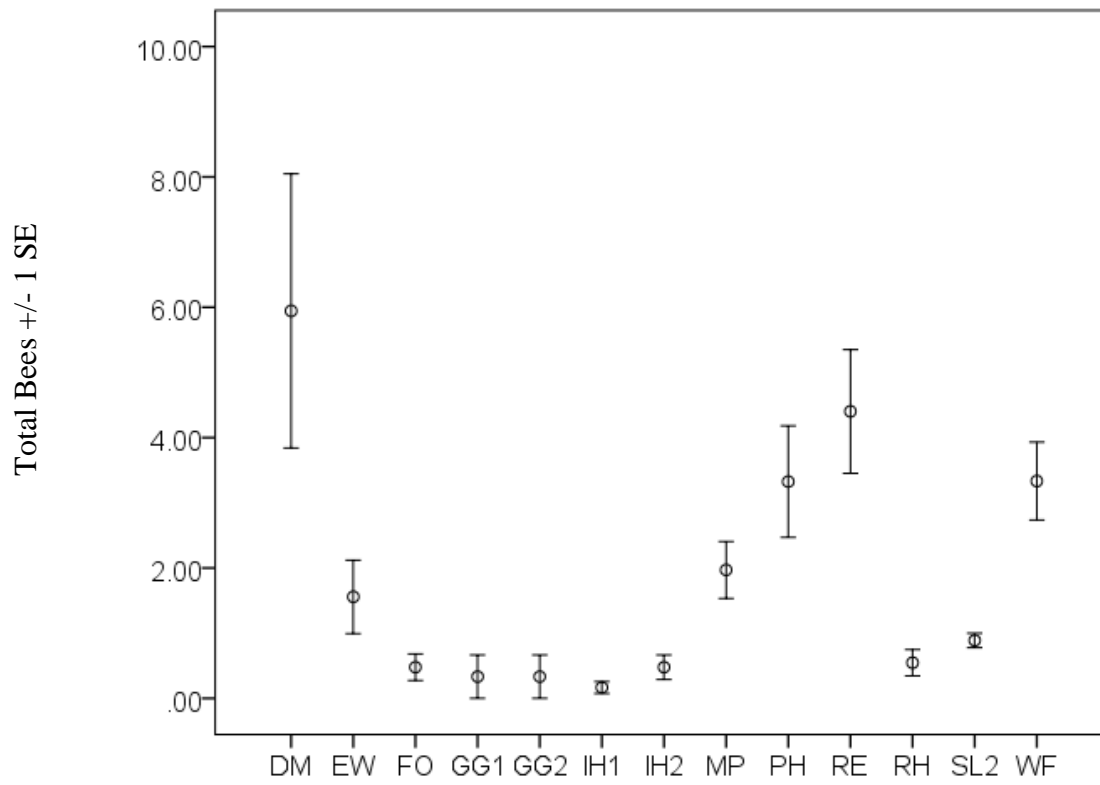


Figure 2 Total bee distribution from March-September 2011 in the 13 study areas

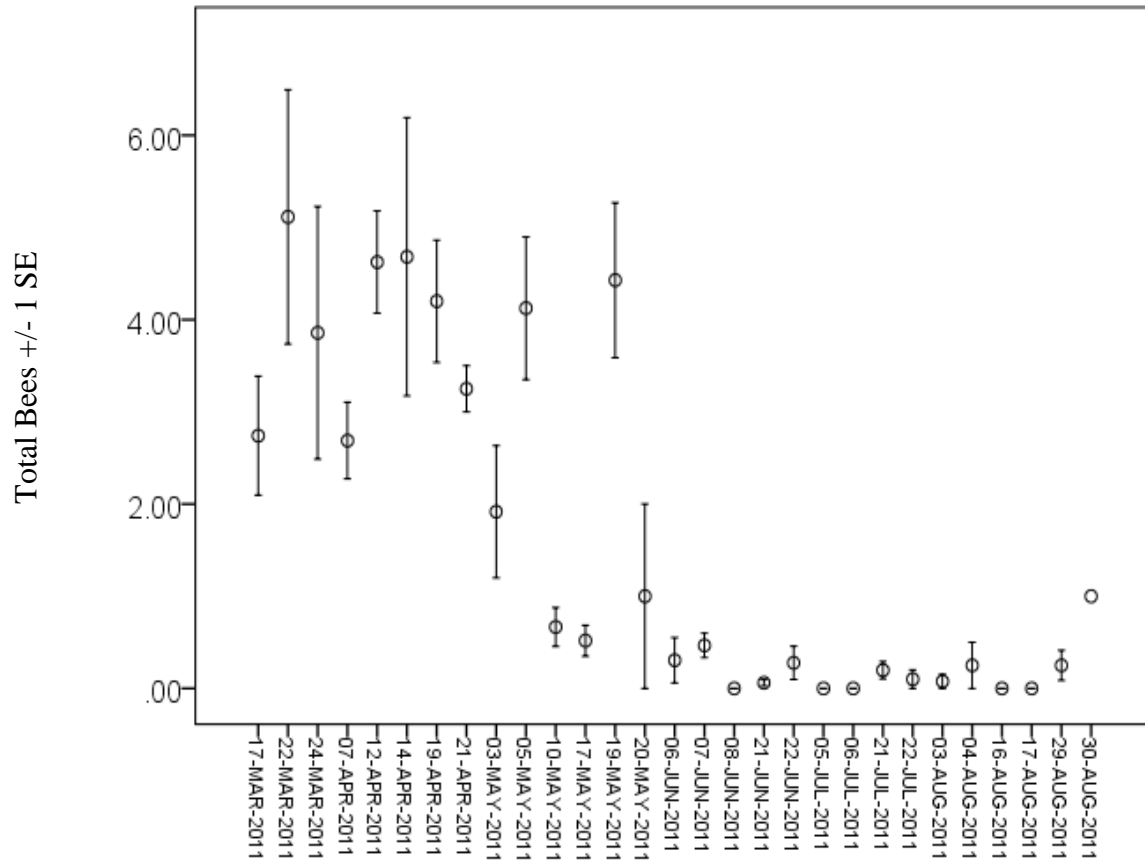


Figure 3 Total bee temporal distribution from March-September 2011

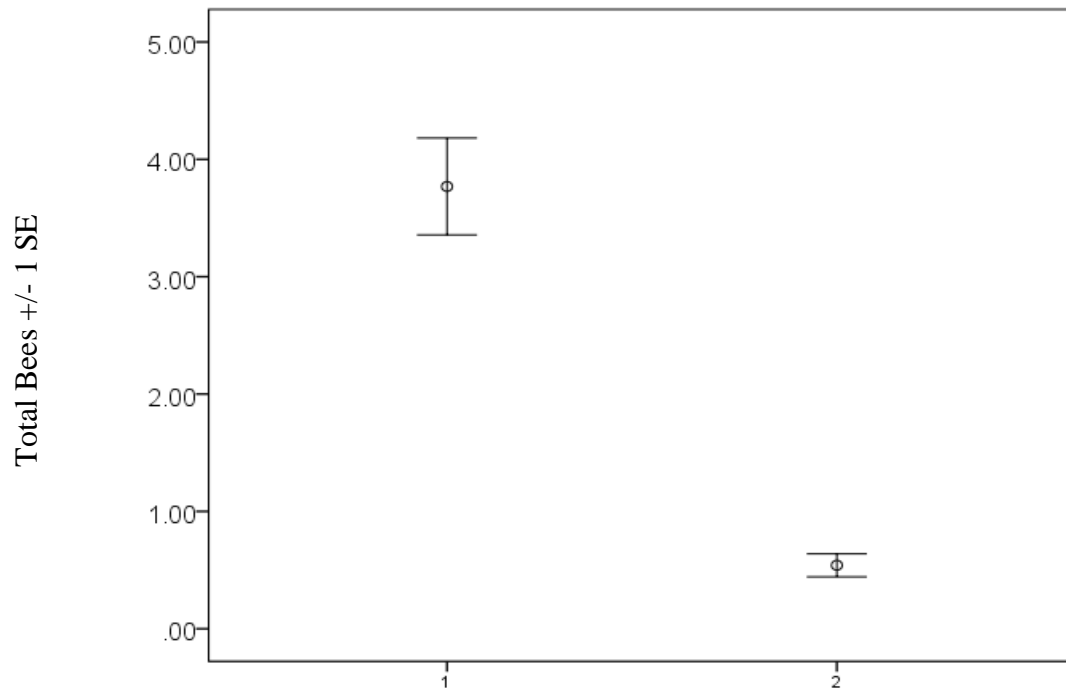


Figure 4 Total bee distribution from March-September 2011 grouped before and after May 1st

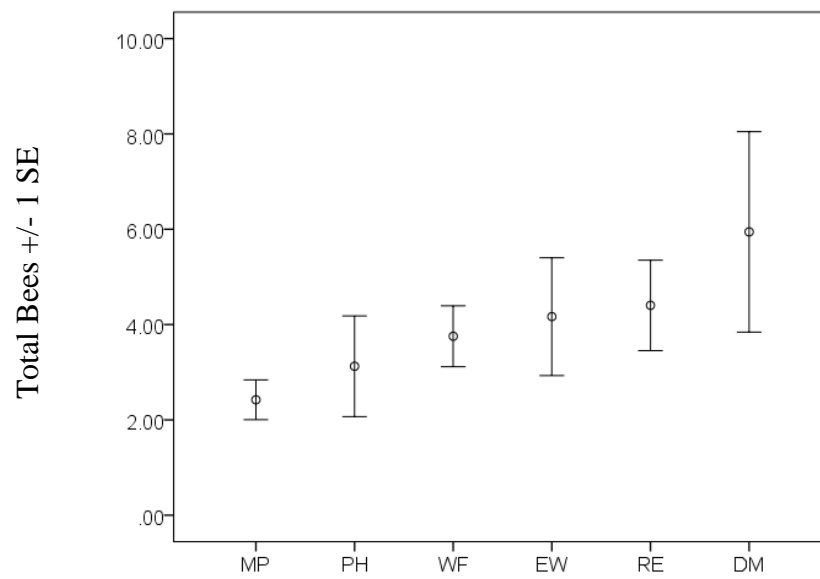


Figure 5 Total bee distribution per site from March-May 1<sup>st</sup> 2011

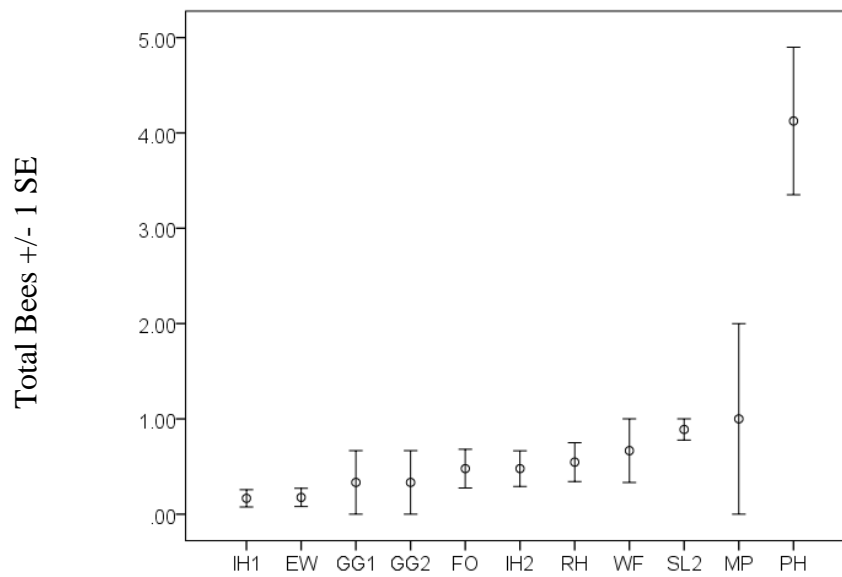


Figure 6 Total bee distribution per site from May 1<sup>st</sup> – September 2011

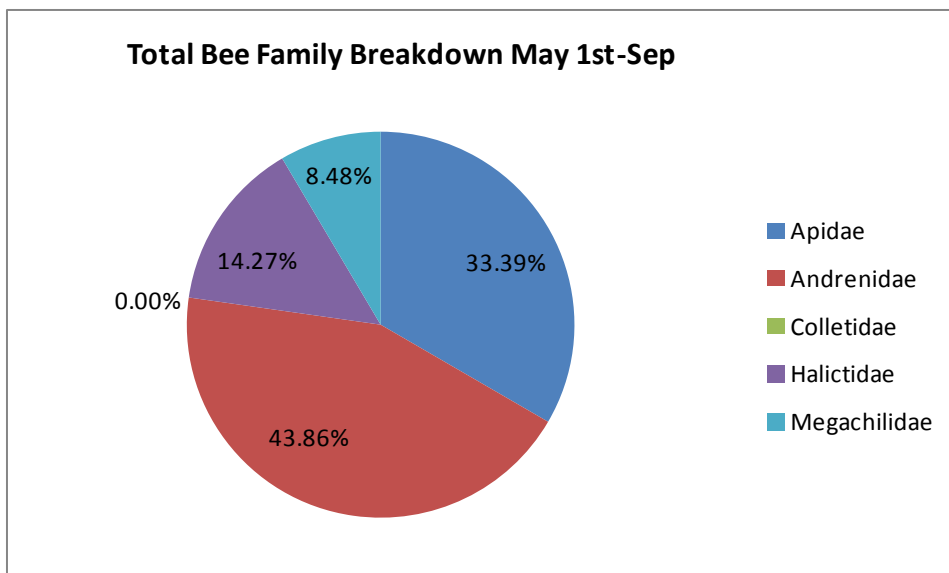
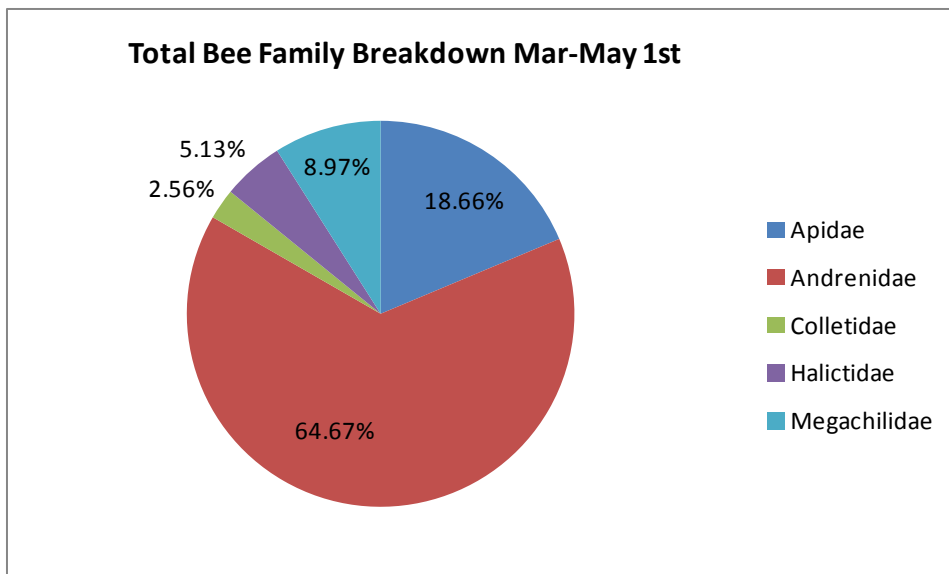


Figure 7 Family breakdown of total bees collected before and after May 1<sup>st</sup> 2011

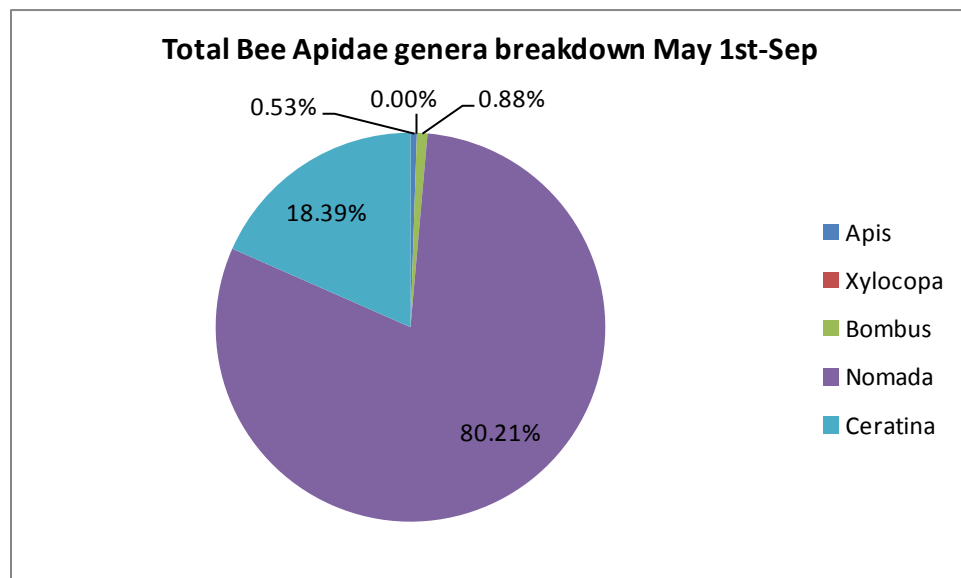
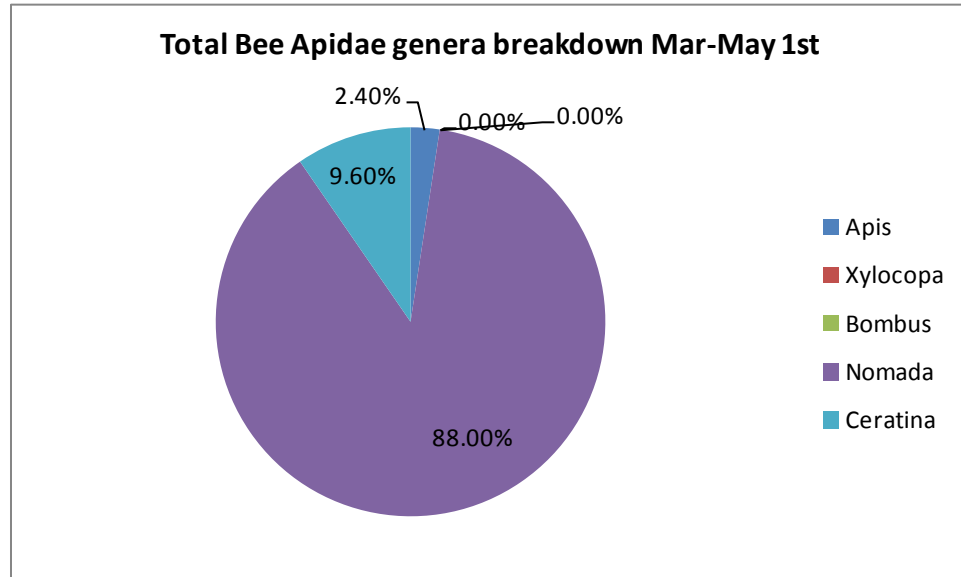


Figure 8 Genera breakdown for total Apidae collected before and after May 1<sup>st</sup> 2011

Table 2      Shannon Diversity Index results for bee families in 5 sites March-September 2011

	Diversity	Family Richness	Evenness
Ecology Woods	0.318	4	0.343
Folk	0.497	4	0.409
Iron Hill 1	0.217	2	0.62
Iron Hill 2	0.498	4	0.41
Rittenhouse	0.355	3	0.474

Table 3      Shannon Diversity Index results for Apidae genera in 5 sites March-September 2011

	Diversity	Genera Richness	Evenness
Ecology Woods	0	1	1
Folk	0	1	1
Iron Hill 1	0	1	1
Iron Hill 2	0.343	3	0.469
Rittenhouse	0.09	2	0.547

Table 4 Significant and approaching significant regression results before May 1<sup>st</sup>

Regression	p-value	r <sup>2</sup>	β	Conclusion
Total Bees vs. Multi-flora Rose Ground Cover	0.0608	0.0443	-0.7394	approaching significance
Total Bees vs. Human Development	0.0656	0.0428	-0.0107	approaching significance
Total Bees vs. Patch Size	0.0282	0.0602	1.5056	significant
Apidae vs. Patch Size	0.0229	0.0646	0.0128	significant
Apidae vs. Human Development	0.0040	0.1012	-1.9011	significant
<i>Apis</i> vs. Patch Size	0.0352	0.0556	0.0006	significant
<i>Apis</i> vs. Non-Native Ground Cover	< 0.0001	0.1946	0.0012	significant
<i>Nomada</i> vs. Patch Size	0.0306	0.0585	-2.0795	significant
<i>Nomada</i> vs. Human Development	0.0011	0.1283	-2.0795	significant
<i>Nomada</i> vs. Impervious Surface	0.0331	0.0569	-2.0337	significant
<i>Nomada</i> vs. Multi-flora Rose Ground Cover	0.0782	0.0392	-0.0169	approaching significance
Colletidae vs. Patch Size	0.0124	0.0774	0.0035	significant
Colletidae vs. Non-Native Ground Cover	<0.0001	0.1810	0.0057	significant
<i>Nomada</i> vs. Colletidae	0.0336	0.0566	0.0611	significant
<i>Nomada</i> vs. <i>Apis</i>	0.0352	0.0556	0.0119	significant

Table 5 Significant and approaching significant regression results after May 1<sup>st</sup>

Regression	p-value	r <sup>2</sup>	β	Conclusion
Total Bees vs. Human Development	0.0208	0.0290	0.0272	significant
Total Bees vs. Impervious Surface	0.0405	0.0228	0.0124	significant
Apidae vs. Patch Size	0.0611	0.0191	-0.0003	approaching significance
Apidae vs. Human Development	0.0451	0.0219	0.6133	significant
Apidae vs. Impervious Surface	0.0320	0.0250	1.2811	significant
<i>Ceratina</i> vs. Human Development	0.0003	0.0706	0.6979	significant
<i>Ceratina</i> vs. Impervious Surface	0.0001	0.0773	1.4254	significant
Andrenidae vs. Human Development	0.0336	0.0246	0.4417	significant
Halictidae vs. Impervious Surface	0.0528	0.0204	-0.3096	approaching significance
Megachilidae vs. Human Development	0.0317	0.0251	0.1559	significant
Vespidae vs. Patch Size	<0.0001	0.0824	0.0011	significant
<i>Nomada</i> vs. Andrenidae	<0.0001	0.1700	0.3686	significant
<i>Nomada</i> vs. Megachilidae	<0.0001	0.1193	0.1078	significant

Table 6 Significant regression results for time independent variables

Regression	p-value	r <sup>2</sup>	β	Conclusion
Patch Size vs. Multi-flora Rose Ground Cover	0.0003	0.0700	-0.0015	significant
Patch Size vs. Non-Native Stems	<0.0001	0.1190	-0.0173	significant



## **Chapter 4**

### **DISCUSSION**

When reviewing the collection data, precipitation may play a key role in discrepancies for some species' presence/absence and abundance. According to data collected from the University of Delaware weather station on the Ag Farm, the average normal rainfall (from the years 1981-2010) was 29.46 inches from March to September (DGS 2012). Summer 2011, notably the month of August, was extremely rainy with an average rainfall from March to September of 36.13 inches – an average 6.67 inches above normal. August 2011 alone had an average rainfall of 14.23 inches compared to the 3.96 inch average (DGS 2012).

#### **Native Bee Compositions**

Overall, the native bees collected from March to September 2011 provide an interesting sampling of the order Hymenoptera and family Apidae in northeastern Delaware forest fragments. The significance of the temporal differences, as expected, provides interesting information about native bee life cycles as their numbers decline towards the end of the summer. The inventory reveals what families and genera are not represented and calls for further research and more focused future studies to better understand native bee species richness, frequency, and distribution in fragmented ecosystems.

The pie charts in Figures 7 and 8 depict the composition of native bees in Newark's forest fragments. Bees from the family Andrenidae comprised a large majority of the collected bees. Bees from this family generally have an early spring

emergence which starts to decrease by June. The genera *Nomada* represented a vast majority of the Apidae collected both before and after May 1<sup>st</sup> with relatively similar percentages – 88% and 80.21% respectively; however, little is known about how their top-down cleptoparasitic behavior affects their solitary bee hosts (Bohart 1970, Kremen et al. 2007). No *Xylocopa* bees were collected in bee bowls throughout the entire course of the collection period and very few were netted.

Unfortunately, *Bombus* specimens were rarely collected – in bowls or netting. However, the time constrained netting in this study was not completed consistently until the end of May due to inclement weather, possibly causing a failure in detecting and collecting *Bombus* early on in the season. *Bombus* spp. are early season bees as their 20-week colonies usually die between August and October (Heinrich 1979). Although native bumble bees were once common in the United States, most of the bee species currently listed as declining are *Bombus* species (Shepherd et al. 2005). There are four *Bombus* species listed on the Xerces Society for Invertebrate Conservation's Red List for at-risk North American pollinators (National Research Council 2007). According to recent literature, *Bombus* species are negatively affected by the loss of native vegetation (Winfree and Kremen 2008). However, pollen collecting bees do increase their foraging distance in simple landscapes, especially during the month of June (Steffan-Dewenter & Kuhn 2003), and large-bodied bees were found to rank least abundant in forest landscapes as opposed to suburban/urban and agriculture in Winfree et al. (2007b). These two factors (and collection problems) may explain their absence from the collection though *Bombus* may still utilize the forest fragments as nesting sites.

In a recent study in Delaware, it was found that agricultural fields such as cucumber, watermelon, and pumpkin had almost double the number of native bees per transect in bee bowls than did state forest, state park, wildlife areas, or organic farms (Keuhn & Disque 2010). The majority of the bee numbers for the 2010 study were collected during the months of July and August, and the distribution of bees differed greatly from the bees sampled in this forest fragment study. Halictidae comprised 72%, Apidae 24%, Megachilidae 2%, and other at 2%, whereas Andrenidae dominated the composition of the native bees collected in the current forest fragment study. The vast majority of all bees collected in the 2010 study were of the Halictidae genus *Lasioglossum*. Apidae composition was not too different compared to the 18.66-33.39% composition in this study, though the Apidae genera included many more: 8 species of *Bombus*, *Ceratina*, *Eucera*, *Melissodes*, *Nomada*, 1 species of *Xylocopa* (*Xylocopa virginica*), *Svastra*, *Ptilothrix*, and *Peponapis*. Only 1 genus of Andrenidae was collected - *Perdita* which was far less diverse than the Andrenidae in the current forest fragment study. Because forest fragments typically offer more undisturbed ground nesting sites and patchy resource clumps compared to the monoculture agricultural fields with endless forage, there are bound to be different proportions and genera of native bees found. Though not too surprising because they are wood nesters, *Xylocopa* barely contributed to the native bee composition and only one species was found in the 2010 study (Keuhn & Disque 2010).

### **Diversity Index**

For total bee family diversity, Iron Hill-2 had the highest diversity in the index of the 5 most consistently studied sites. Folk had the second highest diversity index. Iron Hill-2 was the only site with a reasonable diversity of Apidae members. Iron Hill-

2 had the lowest surrounding impervious surface with 0.91%, no surrounding agriculture, the lowest surrounding human development with 13.68%, relatively high total ground cover with 25.26% of which it had the highest amount of non-native ground cover with 13.85%. Multiflora rose comprised 0% of the ground cover composition.

Folk had increased surrounding impervious surface with 18.83%, no surrounding agriculture, increased human development with 42.34%, much less total ground cover with 5.39%, much less non-native ground cover with 0.62% and 0% multiflora rose composition. Though many of the development parameters were very different compared to Iron Hill-2, the calculated diversities from the index were extremely similar – 0.498 and 0.497 respectively.

Before the study was completed I expected Ecology Woods to have the most native bee diversity. However it ranked as 4<sup>th</sup> out of 5 sites for total bee families and had a diversity of 0 for Apidae collected. Taking the parameters into consideration, Ecology Woods had the highest surrounding agriculture, human development, total ground cover and multiflora rose composition (32.04%, 56.80%, 30.69%, and 3.53% respectively) when compared to the 5 other sites used in the Shannon Diversity Index. When comparing Folk and Iron Hill-2 it was noted that increasing total ground cover and either non-native ground cover or human development (with no agriculture) seemed to create higher total native bee family diversity. Ecology Woods perhaps has too high a degree of disturbance both surrounding the patch and within patch vegetation. The large percentage of intensified agricultural fields and the forage they provide may prevent the bees from utilizing Ecology Woods for anything but nesting sites.

### **Habitat Modification Influences**

According to the results, native bees may utilize human disturbed habitats more than previously thought due to a temporal vegetation inhibitor as the forest canopy leafs out during the spring, aligning the study results with Kremen et al. 2007 and Cane 2001.

Though the regression for total bees against human development within a 100 m buffer of the study areas approached a negative significance before May 1<sup>st</sup>, there was a significant positive relationship found after May 1<sup>st</sup>. Apidae, which includes the well known *Apis* generalist, remained relatively strongly ( $r^2 = 0.1012$ ) and negatively affected by human development before May 1<sup>st</sup> but positively affected by increasing human development after May 1<sup>st</sup>. Because suburban and human disturbed environments provide a range of ornamentals and exotic flowering plants that bloom at different times than do native forest vegetation, native bees may actually travel further distances to forage on these food sources. However, the presence of forest fragments cannot be overlooked in their importance as vital nesting sites for many of the native bees that are either fossorial nesters or above ground nesters in woody vegetation despite the fact that it was not a focus of this study (Cane 2001).

Contrary to what I predicted, there was a temporal difference in patch size's affect on bee abundance. As patch size increased before May 1<sup>st</sup> total bee abundance increased as well, as expected. However, Apidae abundance increased with increasing patch size before May 1<sup>st</sup> but approached significance with decreasing abundance as patch size increased after May 1<sup>st</sup>.

These results align with the idea that though fragmentation may be deleterious for some species, the generalization cannot necessarily be made for all species,

especially aerial insects which may trend towards the disturbance and thrive due to the presence of both forage and nesting sites nearby (Samways 1994, Cane 2001).

Interestingly, agriculture within a 100 m buffer of the study area was not found to be significant when compared against bee abundance. The research on native bees' response to agriculture is inconsistent with Kremen et al. (2007) concluding that intensified agriculture has a negative effect on native bee populations due to the destruction of nesting sites and Winfree et al.(2009) finding that while anthropogenic change negatively affected native bees, agriculture did not. The fact that there was no significance discovered between native bee abundance and agriculture leads me to believe that the forest fragments allowed for available nesting sites even if the intensified surrounding agriculture destroyed others. Thus, according to Winfree et al. (2007a), a high dispersion of natural habitat fragments may support a high biodiversity of bees even if the total natural habitat composition is relatively low.

The results of the time independent regression of patch size vs. multiflora rose ground cover composition and non-native stems reinforce the idea behind Faulkner (2004) that as patch size decreases and edge increases, the encroachment of exotic and invasive species increases as well. As patch size increased, both multiflora rose ground cover composition and the percent composition of non-native stems decreased.

While only the total bee abundance approached a significant negative relationship with multiflora rose composition before May 1<sup>st</sup> (when multiflora rose blooms), both *Apis* and Colletidae abundance increased as the percent of non-native ground cover increased before May 1<sup>st</sup>. *Nomada* abundance significantly decreased as multiflora rose composition increased. Colletidae, a solitary bee family, and *Apis*, a generalist pollinator, have been observed visiting multiflora rose plants (Jesse et al.

2006) and thus utilizing the forage provided by non-native vegetation. However, it remains surprising that this relationship existed before May 1<sup>st</sup> when more native vegetation was in bloom. Preferences for different forage cannot be extrapolated from the data as it was not a focus for the study; however, future projects would be enhanced with this added information as the multiflora rose and other non-native vegetation encroachment and establishment in the forest fragments is largely understudied as it affects native bee behavior.

The genus *Nomada* in itself is an incredible member of the Apidae family. Its cleptoparasitic behavior and effect on its host populations is still largely undocumented. Though parasitism rates were not a component of this study, it is clear from the native bee composition charts that *Nomada* did make up an extremely large percentage of the specimens collected, especially of Apidae.

*Nomada* abundance significantly declined as patch size increased before May 1<sup>st</sup> with no significant relationship after May 1<sup>st</sup>. Unexpectedly, as surrounding human development and impervious surface increased, *Nomada* abundance decreased; as multiflora rose composition increased, *Nomada* abundance approached a significant decrease. According to the literature, I predicted that as the degree of disturbance increased, so too would the solitary bee hosts' vulnerability to increased rates of parasitism. However, increased *Nomada* abundances do not necessarily correlate with increased rates of parasitism. Perhaps the smaller the fragment, and the better the habitat quality, allows for more efficient nest detection and parasitism by *Nomada* species.

The trends found between native bees, surrounding disturbance and vegetation composition in forest fragments in Newark, Delaware provides fodder for further,

more focused studies. The successful inventory of native bee families and their compositions will allow for a solid baseline for future research and has allowed for future methodology alterations such as more consistent early sampling (both bee bowls and time constrained netting) in a greater number of FRAME sites. Focusing on specific relationships between bees and native vegetation, specifically flowering times in the forest fragments, would provide vital information to see if native bees actually prefer non-native vegetation forage or do they simply visit it because that is the forage that is available. Recording temporal vegetation blooms and plant identifications may be necessary to pinpoint these relationships and to see if multiflora rose specifically inhibits native bee foraging behavior. Additionally, *Nomada* parasitism's effect on its solitary bee host populations would generate many interesting studies within the FRAME project and greatly enhance the literature on its unique life history.



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