4-30-2021

The effect of roadside mowing and road traffic on bumble bee abundance and pollinating insect habitat quality in New York highway rights-of-way.

Alyssa Schoenfeldt
axs9530@rit.edu

The effect of roadside mowing and road traffic on bumble bee abundance and pollinating insect habitat quality in New York highway rights-of-way.

Alyssa Schoenfeldt

A thesis submitted in partial fulfillment of the requirements of the degree of Master of Science in

Environmental Science

Thomas H. Gosnell School of Life Sciences

College of Science

Rochester Institute of Technology

Rochester, NY

April 30th, 2021

Committee Approval

Kaitlin Stack Whitney May 13, 2021

Thesis Advisor

Elizabeth Hane May 12, 2021

Committee Member/Advisor

Nathan Lee May 13, 2021

Committee Member/Advisor

Permission for Reproduction

Table of Contents

Table of Figures

Table of Tables

Acknowledgments

This report was funded in part through grant(s) from the Federal Highway Administration and United States Department of Transportation under the State Planning and Research Program, Section 505 of Title 23, U.S. Code provided by the New York State Department of Transportation. The contents of this report do not necessarily reflect the official views or policy of the United States Department of Transportation, the Federal Highway Administration, or the New York State Department of Transportation. This report does not constitute a standard, specification, regulation, product endorsement, or an endorsement of manufacturers. Additional funding came from RIT COS Emerson fellowship.

I would like to extend my thanks to Virginia Aswad, Debmalya Ray Choudhuri, Shereef Ghoneim, Jacob Horn, Alyssa Juergens, Matthew Madsen, Rachel Taylor, Amanda Van, Thuy Huong Vu, Jay Wickard, and Andrew Yoder for helping with the project throughout the last two years.

I would like to thank my committee members, Dr. Elizabeth Hane, Dr. Nathan Lee, and Dr. James Winebrake for assisting me in this project.

I am immensely grateful for Dr. Kaitlin Stack Whitney continually supporting me throughout my academic career and through the process of writing my thesis.

And, finally, I would like to thank my friends and family that have directly or indirectly helped me with my thesis and throughout my college experience by creating a supportive living environment.

Abstract

Many insect pollinators, including native pollinators such as bumble bees (*Bombus* spp.), are facing population declines globally due to loss of natural habitats and other anthropogenic factors. The mandated grassy areas alongside roads, known as roadside rights-of-way (ROWs), are potential habitats for insect pollinators. Yet, roadsides ROWs are highly susceptible to disturbances including on-road traffic and roadside management practices, such as mowing, that may impede their performance as suitable habitat. My research objective was thus to examine if and how road traffic levels and roadside mowing interact to influence pollinating insect habitat quality and bumblebee abundance in highway roadside ROWs across New York State. I tested this using a variety of field survey methods in 2019 and 2020, along 30 highways (n=177 sampling locations) with Control Mowing- Low Traffic (n=33 sampling locations), Control Mowing - Medium Traffic (n=29), Control Mowing - High Traffic (n=27), Modified Mowing - Low Traffic (n=28), Modified Mowing - Medium Traffic (n=35), and Modified Mowing - High Traffic (n=25) treatments. Using generalized linear mixed models, I found no significant difference in habitat quality for pollinating insects between treatments. I was unable to quantitatively assess the treatment effect on bumble bee abundance, due to the extremely limited number of bumble bees observed (98% of n=916 observations across two methods and two years found 0 Bombus spp.). Further research is needed to know if and how roadside ROWs can support insect pollinators, including bumble bees.

Chapter 1: Changes in roadside mowing and road traffic level are not associated with differences in habitat quality for insect pollinators along highways in New York State

Abstract

Insect pollinators are critical to the maintenance of global pollination and biodiversity, but these services are threatened due to rising anthropogenic influences such as degradation and removal of habitat. The loss of habitat poses one of the biggest threats to insect pollinator populations, causing alternative habitats, such as roadside rights-of-way (ROW), are being explored as a possible replacement for lost habitats. Yet, roadsides ROWs are highly susceptible to disturbances including on-road traffic and roadside management practices, such as mowing, that may impede their performance as insect pollinator habitat. My research objective was thus to examine if and how roadside mowing and road traffic levels interact to potentially influence pollinating insect habitat quality in highway roadside ROWs across New York State. I tested this using three habitat quality methods in 2019 and 2020, along 30 highways (n=177 sampling locations) with six treatments: Control Mowing- Low Traffic (n=33 sampling locations), Control Mowing - Medium Traffic (n=29), Control Mowing - High Traffic (n=27), Modified Mowing - Low Traffic (n=28), Modified Mowing - Medium Traffic (n=35), and Modified Mowing - High Traffic (n=25). I used negative binomial, zero inflated beta, and zero inflated negative binomial generalized linear mixed models and estimated marginal means comparisons as post hoc tests of these associations. I found no significant difference in habitat quality for pollinating insects among treatments. However, visit and sampling year were positively associated with several of the measures of habitat quality, potentially indicating the role of long-term vegetation

management and interannual variation on habitat quality. Continued consideration into how roadside habitats could be improved for insect pollinators will prove to be valuable in efforts to support insect pollinator conservation.

Introduction

Insect pollinators are essential to the maintenance of ecosystem services and support the biodiversity of plants through pollination services. The mutualistic relationship between pollinators and flowering plants further promotes biodiversity by aiding in plant reproduction. However, pollinating insect populations, especially wild pollinators, are decreasing worldwide (Cameron et al., 2011; Hopwood, 2008; Vanbergen et al., 2013).

Insects are some of the most important pollinators to plants worldwide. Pollination is performed by a wide variety of different insects such as flies, wasps, solitary and social bees, beetles, moths, butterflies (Vanbergen et al., 2013: Wojcik & Buchmann, 2012). Insect pollinators are critical to the pollination of crops and wildflowers as well as natural pest control measures (Phillips et al., 2020; Vanbergen et al., 2013; Volenec & Dobson, 2019). Each type of insect pollinator is unique in its life history, habitat preferences, plant preferences, climatic conditions, and ability to respond to changes in the environment.

A key difference in these preferences comes from whether the insect pollinator is a wild or a managed species. The most prominent managed insect pollinator species is the European Honey Bee (*Apis mellifera*). Honey bees are not native to North America, but they have become a key species in food crop pollination (Winfree et al., 2009; Wood et al., 2020). Honey bees are also social insects that live in large colonies that are often managed by humans within agricultural environments, and they have been facing colony loss globally (Buri et al., 2014). Additionally, disease spread within colonies and the application of pesticides, notably neonicotinoids recently, in agricultural environments further threaten the ability of honey bees to perform pollination services (Russell et al., 2005; Vanbergen, 2013).

Wild insect pollinators are similarly threatened by disease and pesticides. In agricultural environments, pesticides and herbicides are commonly used to control unwanted plant species as well as other plants in the vicinity. While land managers recognize the value of insect pollinators, pesticides are still applied: pesticide and herbicide management plans are typically created with the intention of not causing harm to honey bees because of their role in crop pollination (Vanbergen et al., 2013). These quasi-protections afforded to honey bees do not always translate to wild bees. Pesticides can have sublethal effects on wild pollinators, contributing to their decline in population (Russell et al., 2018; Vanbergen et al., 2013; Wood et al., 2020). Similarly, wild insect pollinators are susceptible to their own host of diseases. Interestingly, wild insect pollinators are put at risk by the escape of managed populations of bumble bees (*Bombus spp.*) and honey bees into the natural environment (Colla, 2016; McNiel et al., 2020). Wild bees have not had prior exposure to these pathogens and will not have adapted to or created a tolerance to them in the same way that honey bees, that carry the disease, have been able to. This can leave wild bees susceptible to large population declines.

Threats to wild insect pollinators, because of the diversity of morphospecies, are susceptible to additional threats. Threats to wild insect pollinators include competition, climate change, invasive species, and loss/degradation of habitat due to human development (McNeil et al.,2020; Ogilvie et al., 2017; Thomson, 2016; Vanbergen et al., 2013; Williams et al., 2011). In contrast to honey bees, wild insect pollinators are primarily threatened by ecological disturbances (Russell et al., 2005). One of these ecological disturbances is competition with other insect pollinators for resources. Wild bees can be outcompeted by other pollinators such as honey bees (Thomson, 2016). Honey bees live in large colonies and are generalists, meaning they are able to use a wider array of resources in the environment (Colla, 2016; Thomson, 2016).

The presence of honey bees has been linked to an absence of bumble bees (*Bombus spp.*), likely due to their ability to successfully compete for floral resources and the introduction of pathogens to wild populations (Colla, 2016; Thomson, 2016). Native insect pollinators often prefer or specialize in the pollination of specific, native plants (Kasten et al., 2016; Russell et al., 2018).

Wild insect pollinator populations are further impacted by changes to their floral resources, habitat, and climate. Urbanization and development remove natural areas that were previously habitat for plants and animals (Ogilvie et al., 2017; Vanbergen et al., 2013). Insect pollinators and the plants they pollinate rely on each other for their survival, so the removal of one can harm the other. Loss or removal of flowering plants has been observed to lead to decreased abundance of wild bees (Hopwood, 2008; Nichols et al., 2019; Ogilvie et al., 2017; Thomson, 2016). The area needed and used for ground nesting is also lower in urban environments (Ahrné et al., 2009). Human changes to the environment in the name of development have contributed to global climate change. Climate change has led to changes in phenological and geographical ranges of plants and pollinators (Bartomeus et al., 2011; Colla, 2016; Ogilvie et al., 2017; Pyke et al., 2013). The changes in the ranges for plants and pollinators are often not occurring at the same rate or in the same direction (Bartomeus et al., 2011; Colla, 2016). Climate change, along with other changes to the plant community, can harm both pollinators and plants due to overall decreases in pollination.

The removal and degradation of habitat remains one of the biggest threats to insect pollinator populations and habitat. This has encouraged researchers to further investigate if and how populations can use semi-natural and developed environments as habitat (Ahrné et al., 2009; Leonard et al., 2018; Leston et al., 2019; Winfree et al., 2009). A potential developed area that researchers are suggesting as alternative habitat are rights-of-ways (ROWs). ROWs are linear corridors for transportation infrastructure (utility lines, railways, roads). The areas next to the infrastructure are often vegetated, creating natural to semi natural areas with resources that may be missing from the environment (Gardiner et al. 2018; Villemey et al., 2018).

Highway roadside ROWs are potential habitats that may contain vegetation that is absent in the surrounding developed landscapes. Roads and their associated ROWs are ubiquitous: there are over four million miles of road in the United States (U.S. Department of Transportation, 2019) and an estimated 3,903,722 hectares for potential roadside habitat (Wojcik & Buchmann 2012). Roadsides are actively managed grassy habitats that vary in vegetation and width.

Yet roadsides ROWs are highly susceptible to disturbances and degradation that may impede their performance as insect pollinator habitat. Disruptions to roadsides come from both on road and roadside factors. Activity that occurs on the road has the ability to impact insect pollinators in roadside ROWs. In particular, high traffic roads have the greatest ability to degrade the environment because they experience more exposure to noise pollution (Davis et. al., 2018), chemical deposition (Khalid et al., 2018; Muñoz et al., 2015), collision with vehicles (Halbritter et al., 2015; Muñoz et al., 2015; Phillips et al., 2019), and barriers to dispersal (Theodorou, 2020; Muñoz et al., 2015). For example, Monarch butterflies (*Danaus plexippus*) have been observed to be desensitized to stressors, potentially exposing them to danger while traveling between patches of vegetation (Davis et. al., 2018). Chemical deposition of salt and heavy metals from vehicles in roadsides can affect the health and growth of vegetation, namely more sensitive wild flowering plants (Khalid et al., 2018; Muñoz et al., 2015). Because floral resources may not be suitable in a given patch of roadside habitat, insect pollinators may seek out resources in a location separate from their nesting site (Halbritter et al., 2015). This may restrict the areas in which nests are established (Russell et al., 2018) as well as lead to colonies spending more time foraging (Pyke et al., 2011). Road traffic also introduces the possibility of insect pollinators and vehicle collisions. Higher traffic roads have been associated with increased mortality rates of insect pollinators (Halbritter et al., 2015; Muñoz et al., 2015; Phillips et al., 2019). Traffic can pose as a barrier for insect dispersal, isolating insect pollinators from other floral resources and breeding populations (Theodorou, 2020; Muñoz et al., 2015). While dispersal is less of a threat to large-bodied insect pollinators, including bumble bees (McFrederick & LeBuhn, 2006; McNiel et al., 2020; Theodorou et al., 2020), smaller insect pollinators like many wild bees do not have this advantage.

While roads can negatively impact insect pollinators, the effect of traffic on insect pollinator presence and habitat quality for pollinators remains inconclusive. Increased traffic has been associated with increased mortality of butterflies (Skórka et al., 2013), decreased abundance of pollinators (Phillips et al., 2019), and no impact on the abundance of butterflies (Munguira & Thomas, 1992; Saarinen et al., 2005) or dragonflies (Soluk et al., 2011). Additionally, low traffic roads had higher richness of bee forage plants, compared to high traffic roads (Wrzesień & Denisow, 2016), and the same species richness of beetles as in high traffic roadsides (Melis et al., 2010). These varied outcomes may be a result of the roadsides selected having low traffic volume large highways (Phillips et al., 2020; Saarinen et al., 2005; Skórka et al., 2013; Wrzesień & Denisow, 2016), in comparison to ROWs along large highways. A comprehensive understanding of habitat quality and insect pollinator success in ROWs along varied road traffic volume can help determine which sites would be prioritized when considering alternative habitat and conservation.

Simultaneously, disturbances in the roadside may influence whether roadside ROWs can provide sufficient habitat for insect pollinators. ROWs are semi-natural vegetative habitats, but because they are provided for accessibility and transportation purposes, they are actively managed. Insect pollinators are dependent on the presence of floral resources for foraging and habitat, so alterations to these resources have the ability to therefore harm bees as a result. Disturbances in the vegetated areas that can impact insect pollinators include invasive plant species, application of pesticides and herbicides, and intensive mowing practices (McCleery et al., 2015; Muñoz et al., 2015). Generalists are plants and animals that can quickly adapt to changes and disturbances to their environment can have an advantage to less resilient organisms (Bernes et al., 2016). ROWs are linear corridors for humans, but they can also serve as corridors for invasive plant species to spread (Bernes et al., 2016; Lázaro-Lobo & Ervin, 2019). Invasive plant species can interfere with the growth of wild plant species by out competing them for resources. While invasive plant species are still used by insect pollinators, there can still be a preference for native plants (Williams et al., 2011; Wrzesień & Denisow, 2016).

Herbicides are sometimes used in roadsides to control plant populations, particularly invasive plants. Management strategies such as mowing and the use of herbicides can change the plant and animal populations present (Hopwood et al., 2015; Noordijk et al., 2009). Pesticides used to control unwanted insects can have similar resulting changes to the community assemblage. These chemicals can be absorbed into plant tissue, and into nectar and pollen, that is then passed onto pollinators (Vanbergen et al., 2013; Wood et al., 2020). The impacts of both pesticide and herbicide use on bees have been explored, but mainly use honey bees as the focal organism, primarily due to their large role in pollination of crops (Vanbergen et al., 2013; Wood et al., 2020). However, lethal and sublethal effects have been observed in wild bees (Russell et al., 2018; Vanbergen et al., 2013; Wood et al., 2020).

Mowing is a common management technique for roadside ROWs that may also act as disturbance. Mowing is used for both practical and aesthetic purposes: mowing decreases the height of vegetation in the ROW primarily for safety reasons. Vegetation is mowed in order to ensure motorist visibility and to provide a buffer for cars to regain control and avoid collisions. Generally, the interior of roadside ROWs (further from the road) are not mowed as intensively as the edge of the road The use of roadsides as habitat by insect pollinators has been established (Halbritter et al., 2015; Keilsohn et al., 2018; Phillips et al., 2019; Noordijk et al., 2009; Ries et al., 2001; Saarinen et al., 2005; Skórka et al., 2013), but how well they perform as habitats as management changes is still unclear. Insect pollinators require floral resources for foraging and habitat; removal of these resources may decrease the usage of the roadsides as a habitat for these pollinators. Mowing may reduce floral resources and result in restricting habitat and insect pollinator abundance along roadsides (Gardiner et al., 2018; Noordijk et al., 2009). Wild bees in powerline corridors and managed meadows that have not been mowed have been observed to have increased abundance, when compared to their mowed counterparts (Buri et al., 2014; Russell et al., 2018). While these habitats are different from roadside ROWs, they are managed through similar practices. Excessive mowing (>80% of the habitat), particularly during the growing season, removes the needed resources for wild bees (Buri et al., 2014). Similarly, butterflies in reduced mowing roadsides have also been observed to increase in abundance (Halbritter et al., 2015; Leston et al., 2019).

Other management strategies for managing roadside vegetation and providing habitat for insect pollinators is through strategically timed mowing. The timing of mowing and management activities may also affect their ability to serve as habitat for insect pollinators (Halbritter et al., 2015; Kasten et al., 2016; Noordijk et al., 2009). Because insect pollinators each have different life histories, the time at which they are most active can vary. Similarly, plants have varying blooming periods. Understanding the floral resources as well as the specific insect pollinators that are in roadsides can help determine if necessary mowing could be performed outside of the times most critical to pollination. Monarch butterflies rely on milkweed as a larval host, so removal of milkweed could be detrimental to their repopulation (Leston et al., 2019). Conversely, in roadsides that already have milkweed present, Monarch populations could benefit from mowing prior to when they lay their eggs because they prefer to utilize fresh milkweed (Kasten et al., 2016). The growth of new milkweed occurs after mowing. While changes to mowing patterns indicate that this could benefit insect pollinators, knowing what floral resources are present in the roadside as well as the insect pollinators that utilize them could impact the particular management practice that is used.

Existing research on implementing reduced mowing as a management practice differs in temporal and spatial scales. While studies have found mowed roadside habitats have showed fewer pollinators and flowering plants, Phillips et al. (2020) noted that this may not entirely explain how insect pollinators respond to changes in mowing because of small temporal scales in the research done to date (Phillips et al. 2020). These limitations may not be able to adequately explain, or provide conclusive/non-conflicting results, on how changes in mowing patterns impact pollinators, leading to inconclusive or conflicting results. The current understanding of insect pollinators' response to mowing are limited to small spatial scales, lasting just one growing season. Furthermore, much of the research on the topic has occurred in Europe (Noordijk et al., 2009, Phillips et al., 2019, Saarinen et al., 2005). The spatial scale of insect pollinator roadside studies is further narrowed due to using roadside sites that are within a single region or city, restricting the variability of responses in diverse landscapes (Munguira & Thomas, 1992; Phillips et al., 2019; Noordijk et al., 2009; Ries et al., 2001; Saarinen et al., 2005). Differences on spatial and temporal scales may prove to be critical to understanding the ability of reduced mowing to provide quality habitat.

The size of the roadside ROW may also influence habitat quality for pollinators. Higher traffic ROWs are associated with wider roadside environments (Phillips et al., 2020). The current understanding of the role ROW width plays in insect pollinator abundance and diversity is mostly limited to butterflies (Munguira & Thomas, 1992; Saarinen et. al., 2005; Skórka et al., 2013). Wider roadsides have also been observed to promote butterfly abundance (Saarinen et. al., 2005) and richness (Munguira & Thomas, 1992). These benefits may be attributed to greater accessibility to adjoining habitats, allowing for dispersal (Munguira & Thomas, 1992; Skórka et al., 2013). Furthermore, wider roadsides can provide more area for plants to grow, increasing the floral cover present for insect pollinators to use for foraging and habitat. Conversely, narrower ROWs increase the potential edge effects, or influences from the environment outside of the habitat (Volenec & Dobson, 2019). In the case of roadside ROWs, narrow ROWs place insect pollinators closer to the road, and all of its associated disturbances. Phillips et al. (2020) suggested that sites with wider roadsides along low traffic roadsides be prioritized when considering potential conservation efforts. This suggestion, paired with potential benefits from reduced mowing may indicate ideal locations as well as improved management solutions for insect pollinator habitat.

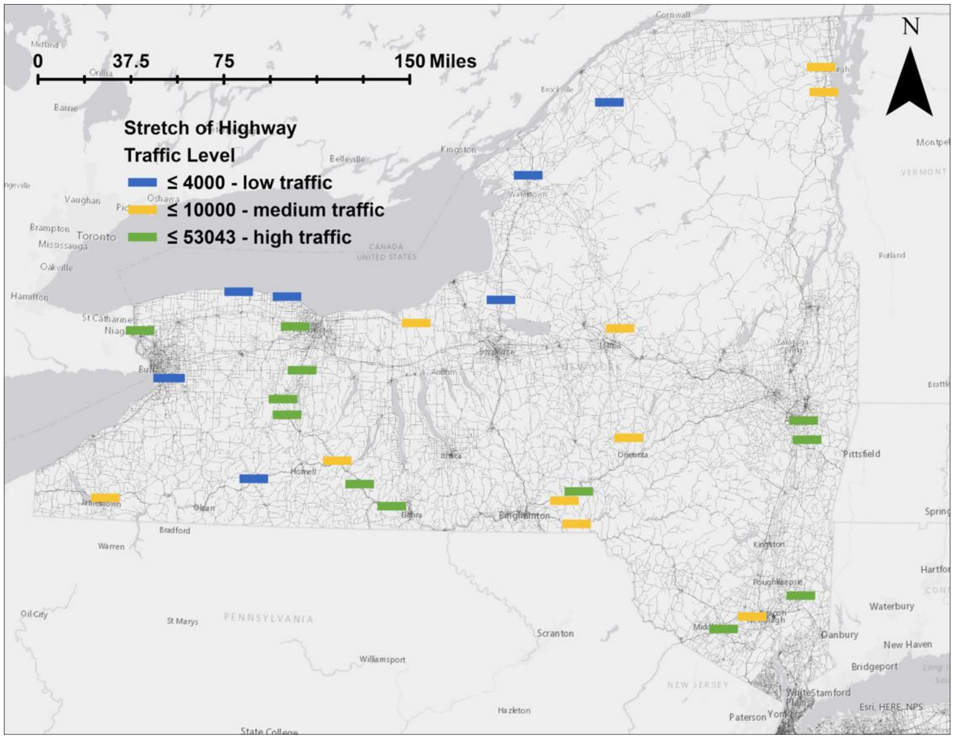
If roadside ROWs are to be locations of quality habitats for insect pollinators, the intersection of road factors, such as traffic, and roadside context, like mowing management, need to be understood more completely. Degradation to interior roadside habitats is heavily influenced by on-road activities, diminishing their ability to serve as quality habitat. Heavy metals and salt from vehicles and pesticides and herbicides applied to vegetation can enter plant tissue and impair its growth and health. Bees are dependent on the presence and quality of floral resources, so these on-road and roadside management practices can have the potential to create environmental conditions that are unsuitable. Roadside mowing and traffic volume may, similarly, interact and create more consequential impacts on habitat quality and bee abundance. The interaction between on-road traffic and mowing practices may help account for unexplained and conflicting data seen in other studies of pollinators within reduced mowing roadside habitats. Currently, roadside mowing and traffic volume are understood as separate factors, but they have not been assessed for their synergistic effect on bees and habitat quality.

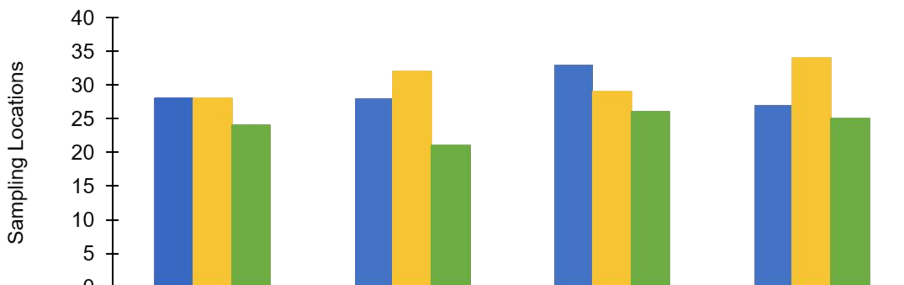
In order to assess the ability of roadside ROWs to serve as quality habitats insect pollinators, I examined the interaction and effects of road traffic and roadside mowing on habitat quality using empirical and remotely sensed data. Specifically, I examined whether there is an interaction between roadside mowing and road traffic volume on roadside insect pollinator habitat quality in New York State (NYS). I hypothesized that areas with intensive mowing management and high road traffic would diminish the quality and suitability of roadside ROWs as habitat. This could be possible due to the increased richness of bee forage plants (Wrzesień & Denisow, 2016), and the abundance of pollinators (Martin et al., 2018; Phillips et al., 2019) in low traffic roadsides. Floral resources can determine the abundance of butterflies and wild bees (Halbritter et al., 2015; Hopwood 2008; Nichols et al., 2019; Ogilvie et al., 2017; Russell et al., 2018; Thomson, 2016), so management practices such as reduced mowing that preserve vegetation in ROWs may be able to connect the relationship between floral resources and road traffic. Furthermore, increases in the abundance of butterflies and wild bees have been observed in unmowed ROWs (Halbritter et al., 2015; Phillips et al., 2019; Russell et al., 2018).

Materials and Methods

Study Sites

This study was conducted along 30 sections of highway in upstate New York to conduct this experiment (Figure 1). These highways were selected to represent diversity of road conditions such as adjacent land use, traffic volume, speed limit, and road size across the state. The sampling locations at each of the highways were established in spring of 2019 and were managed by the NYSDOT through the end of the study period in fall of 2020.





	Control	Modified	Control	Modified
	2019		2020	
Low	28	28	33	27
Medium	28	32	29	34
High	24	21	26	25

Treatment

Low Medium High

Figure 1: This map contains the average daily traffic at each of the stretches of highway (n=30) across New York. Each section of highway had six sampling locations. The graph and table correspond with the number of sampling locations (2019 n=161, 2020 n=174) found across the 30 stretches of highway. The blue dashes represent sampling locations that have average daily traffic less than 4,000 vehicles per day (low traffic). The yellow dashes represent sampling

locations that have an average daily traffic range of 4,001 to 10,000 vehicles per day (medium traffic). The green dashes represent sampling locations that range from 10,001 to 53,043, the maximum, vehicles per day (high traffic). The NYS road network contains all of the roads present in NYS.

Treatments

Each section of highway monitored was composed of four miles of roadside, two miles for each mowing treatment (control and reduced mowing). Within each two-mile section, I established three sampling locations that were separated by at least at least half a mile (Figures 2 & 3). One section of the highway received the control mowing treatment, or the current mowing management (NYSDOT Vegetation Mowing Policy TMI 14-01). This policy indicates that interstates and primary highways should be mowed a single pass of the mower twice a year and that secondary highways be mowed a single pass one a year (New York State Department of Transportation, 2017). The modified mowing treatment was applied to the other section of the highway, mowing the roadside ROW every two years after a plant killing frost. These ROWs were also mowed with a wider pass of the mower, as compared to the control. The NYSDOT did not mow the first year of the study (2019) but mowed later and wider during the second year (2020).

Mowing in both the control and the modified sites occurred beyond the safety strip. The safety strip, by NYS law, is the first 15 feet of the roadside ROW past the pavement that the NYSDOT mows in order to maintain short vegetation (New York State Department of Transportation, 2017). The vegetation is kept short to aid in motorist visibility and to provide space in the event a driver loses control of their vehicle (Figure 2).

Figure 2: A diagram of the mowing treatment at each site. The strip on both sides of the road, as seen in orange, depicts the 15 ft safety strip that is mowed for safety purposes. The green represents the control mowing pattern and blue represents the later and wider, modified mowing pattern.

Figure 3: A diagram with the treatments and their three replicates for highway segment site 26, one of the 30 sections of highway in the study. This is just one example segment to demonstrate the sampling design. Each sampling location is within the treatment's two-mile range and was separated by at least half a mile. The green represents the control mowing, and the blue represents the modified mowing that is wider and mowed less frequently.

Traffic

I used publicly available data from the NYS Department of Transportation (NYSDOT) via the NYS GIS Clearinghouse to utilize counts of annual average daily traffic (AADT) and the road network (New York State Department of Transportation, 2019). NYSDOT uses several techniques to calculate the annual average daily traffic counts. AADT values are calculated from continuous and short counts collected by the Statewide Monitoring System and the Weigh-in-Motion (WIM) Stations and portable traffic counters, (New York State Department of Transportation, 2015). The most recent AADT data available at this point is from the 2019 count statistics (New York State Department of Transportation, 2019).

The AADT values in New York for the entire state ranged from 0 to ~300,000 cars per day. I used ArcGIS Pro 2.7.2 to join the 2019 count statistics table with the 2019 roadway shape file (Esri Inc., 2021; New York State Department of Transportation, 2019). The coordinates of each sampling location were combined with the associated AADT value. I designated the sampling locations into categories determined by their AADT values. I created three traffic categories: low, medium, and high traffic. Low traffic roads had AADT values of 4,000 or fewer vehicles per day, with a minimum value of 23 vehicles per day. Medium traffic roads had between 4,001 and 10,000 vehicles per day. High traffic roads had between 10,001 and the maximum, 53,043 vehicles per day (Figure 1). Of the 177 sampling locations, the mean of the average number of vehicles per day is 9,268 (SD=11,106). Traffic levels in the literature typically do not reach the maximum value as seen in the NYS data set. Because of this, the low, medium, and high traffic levels that I created will correspond to different categorization descriptors used in the literature. I set up my low and medium traffic levels so that they would correspond with many of the intermediate and high levels found in the literature. Many of the traffic ranges found in the literature are primarily along roads that are less than ~10,000 vehicles (McCleery et al., 2015; Melis et al., 2010; Munguira & Thomas, 1992; Phillips et al., 2020; Saarinen et al., 2005; Skórka et al., 2013; Soluk et al., 2011; Wrzesień & Denisow, 2016).

I then used the two mowing treatments (control, modified) and the three traffic categories (low, medium, high) to create six treatments representing interactions of roadside mowing treatment and on-road traffic on the 177 sampling locations (Figure 1). The six treatments included Modified Mowing - Low Traffic (n=27 sampling locations), Modified Mowing - Medium Traffic (n=34), Modified Mowing - High Traffic (n=25) treatments, Control Mowing-Low Traffic (n=33), Control Mowing - Medium Traffic (n=29), and Control Mowing - High Traffic (n=26).

Habitat Quality Assessment

To assess roadside ROWs potential to be suitable habitat for insect pollinators, I used three distinct habitat quality assessments.

To assess habitat quality for insect pollinators generally, I used the *Streamlined Bee Monitoring Protocol for Assessing Pollinator Habitat* (Ward et al., 2014). This protocol uses the abundance and diversity of bees as a proxy for habitat quality. I conducted this assessment twice at each sampling location during both field seasons. The field season ran from May to August, or until all sampling locations were visited twice each season (2019 and 2020). I performed the assessment while walking for 7.5 minutes along a 100 ft transect, located within the ROW, and recording the number of wild bees and honey bees that landed on a flower for at least 0.5 seconds within 3 feet of the transect. I only conducted surveys when environmental conditions were suitable for bee activity. Bees are most active between the hours of 10:00 A.M. and 5:30 P.M, and when ambient temperatures are above 60 degrees Fahrenheit, with wind speeds less than 8 mph, and clear (or partly cloudy/overcast -- 40% cloud cover/you can still see your shadow) (Colla, 2016; Ward et al., 2014). The number of honey bees and wild bees provided a quantitative, indirect measure of habitat quality for insect pollinators.

During the 2020 field season, I also used the *Rights-of-way as Habitat Working Group Pollinator Scorecard* to assess the quality of the roadside as a potential habitat for insect pollinators generally (Rights-of-way As Habitat Working Group, 2019). This assessment is specifically designed for ROW habitats. It was released at the end of August 2019 and thus was not available for the 2019 field season. The scorecard is available in varying tiers of complexity based on the background of the researcher. I used tier three, the most involved tier, involving the identification of plants to the species level. This tier used metrics such as cover of invasive species/noxious weeds, cover of potentially flowering nectar plants, number of (native) nectar plant species, abundance of milkweed, adjacent land use (e.g., developed, diverse/non-diverse grassland, woodland, etc.), and other habitat resources (e.g., native bunch grasses, brush piles, undisturbed thatch, etc.). The presence and/or quantity of these metrics each correspond with a numerical score. The scores for each metric were then summed and correspond to a habitat quality rating (0-20: improvement opportunity, 21-35: basic habitat quality, 36-50 moderate habitat quality, 51-75: high habitat quality, 76+: exemplary habitat quality).

To more specifically assess how roadside ROWs may provide habitat for bumble bees (*Bombus* spp.), I assessed herbaceous forb cover. To do this, I used the line intercept method (100m transect per sampling location) (Kercher et al., 2003). Forbs in particular are critical to bumble bee foraging and habitat (Carvell et al., 2015; Loffland et al., 2017). I quantified the percentage of herbaceous forbs present in the transect, to indicate whether there was abundant foraging and habitat that is required for pollinators. I used this information to examine the amount of vegetation and the families of vegetation present to determine if the habitat had appropriate resources for bumble bees. Bees such as bumble bees have been observed to commonly utilize the Asteraceae and Fabaceae plant families (Ahrné et al., 2009; Nichols et al., 2019; Wrzesień & Denisow, 2016). Plants within the Asteraceae family have been suggested as an estimator of wild bee abundance and diversity (Williams et al., 2001). Asteraceae plants are commonly used due to their high nectar content (Nichols et al., 2019); Wrzesień & Denisow, 2016). Fabaceae plants have nectar with lower sugar content than Asteraceae plants, but their pollen is more protein rich. I conducted this assessment at least twice per sampling location during each field season.

Statistical Analyses

I analyzed the habitat quality of the ROW for each one of the three habitat quality field methods in R statistical software (R Core Team, 2020). Before running analyses, I tested potential independent variables for multicollinearity. Using Pearson's product moment correlations, I removed co-varying variables using a cutoff of correlation coefficients higher than 0.7, correlation coefficients lower than -0.7, or with a p value of <0.05. Based on the results, temperature was the only continuous environmental variable included in all analyses.

In all three of the models, I included temperature (measured in degrees Fahrenheit), site, and sky conditions (categorical variable measured as cloudy, bright overcast, or clear) as random effects. These factors were included because bee activity is highly dependent on weather conditions (Ahrné et al., 2009; Nichols et al., 2016; Theodorou et al., 2020; Thomson, 2016). The six mowing-traffic treatments and year (2019 or 2020) were tested as fixed effects in the streamlined bee and herbaceous forb cover models. I replaced year with visit (1 or 2) for the scorecard model because this data was collected only in 2020.

For the streamlined bee protocol, counts of bees (honey bees, wild bees, and total bees) were the response variables. Due to the high number of zeros observed, I used a zero inflated generalized linear mixed model with a negative binomial distribution using the *glmmTMB* package (Brooks et al., 2017).

For the ROW scorecard, the Tier 3 numerical habitat quality rating was the response variable. I used a generalized linear mixed model with a negative binomial distribution using the *glmmTMB* package. The six mowing-traffic treatments, visit, and ROW width were tested as fixed effects. The scorecard habitat quality rating data was only collected in 2020, so this analysis was on a subset of the dataset. I also scaled ROW width before using it as a fixed effect.

For the herbaceous forb cover, the proportion of the vegetation transect in Asteraceae and Fabaceae was the response variable. I used the *glmmTMB* package to create a generalized linear mixed model with a zero inflated beta distribution. I assessed whether there was an interaction between the mowing-traffic treatments and year by adding an interaction term to the model where they were strictly fixed effects.

After building the models and running analysis of variance tests, I then performed post hoc testing to compare treatment groups using estimated marginal means using the *emmeans* package (Lenth, 2021).

Results and Discussion

Results

Habitat Quality Assessments

I performed 534 Streamlined Bee Monitoring Protocol for Assessing Pollinator Habitat surveys over the course of the two field seasons and 177 sampling locations (Ward et al., 2014). There was a large number of zeros present in the data; 79.03% of honey bee, 76.40% of wild bee, and 65.54 % of total bee observations (n 2019=202, n 2020=332, n total=534) found no bees.

Using analysis of variance, I found that visit was positively correlated (chi square =12.44, $p<0.001$). with wild bee abundance in roadside ROWs (Table 1, Figure 4). The 6 mowing - traffic treatments were not significantly associated with wild bees. The estimated marginal means found that the 6 mowing-traffic treatments did not have any significant differences on wild bees. When the six treatments in 2019 were compared to their 2020 counterpart, there were significant differences present due to time.

Using analysis of variance, visit was positively correlated (chi square =10.82, $p=0.001$) with honey bees in the streamlined bee monitoring protocol (Table 2, Figure 4). The six mowing - traffic treatments were not significantly associated with honey bees. The estimated marginal means found that the six mowing-traffic treatments did not have any significant differences in honey bees. When the six treatments in 2019 were compared to their 2020 counterpart, there were significant differences present due to time.

Using analysis of variance, there were not significant correlations between visit (chi square=0.71, $p=0.40$) and treatment were (chi square=3.89, $p=0.57$) on total bee abundance (Table 3, Figure 4). The estimated marginal means found that the mowing-traffic treatments did not have any significant differences in total bees.

Table 1: Analysis of variance results from the streamlined bee monitoring protocol (wild bees) with the fixed effects year and the roadside mowing - road traffic treatments. I found sampling year to be significantly positively correlated with wild bee abundance.

	Chisq	Df	Pr(>Chisq)
Year	12.44	1	<0.001
Mowing-Traffic Treatment	1.27	5	0.94

Table 2: Analysis of variance results from the streamlined bee monitoring protocol (honey bees) with the fixed effects year and the roadside mowing - road traffic treatments. I found sampling year to be significantly positively correlated with honey bee abundance.

	Chisq	Df	Pr(>Chisq)
Year	10.82	1	0.001
Mowing-Traffic Treatment	5.33	5	0.38

Table 3: Analysis of variance results from the total bees from the streamlined bee monitoring protocol with the fixed effects year and the roadside mowing - road traffic treatments. I found that neither year nor treatment was significantly associated with total bee abundance.

	Chisq	Df	Pr(>Chisq)
Year	0.71	1	0.40
Mowing-Traffic Treatment	3.89	5	0.57

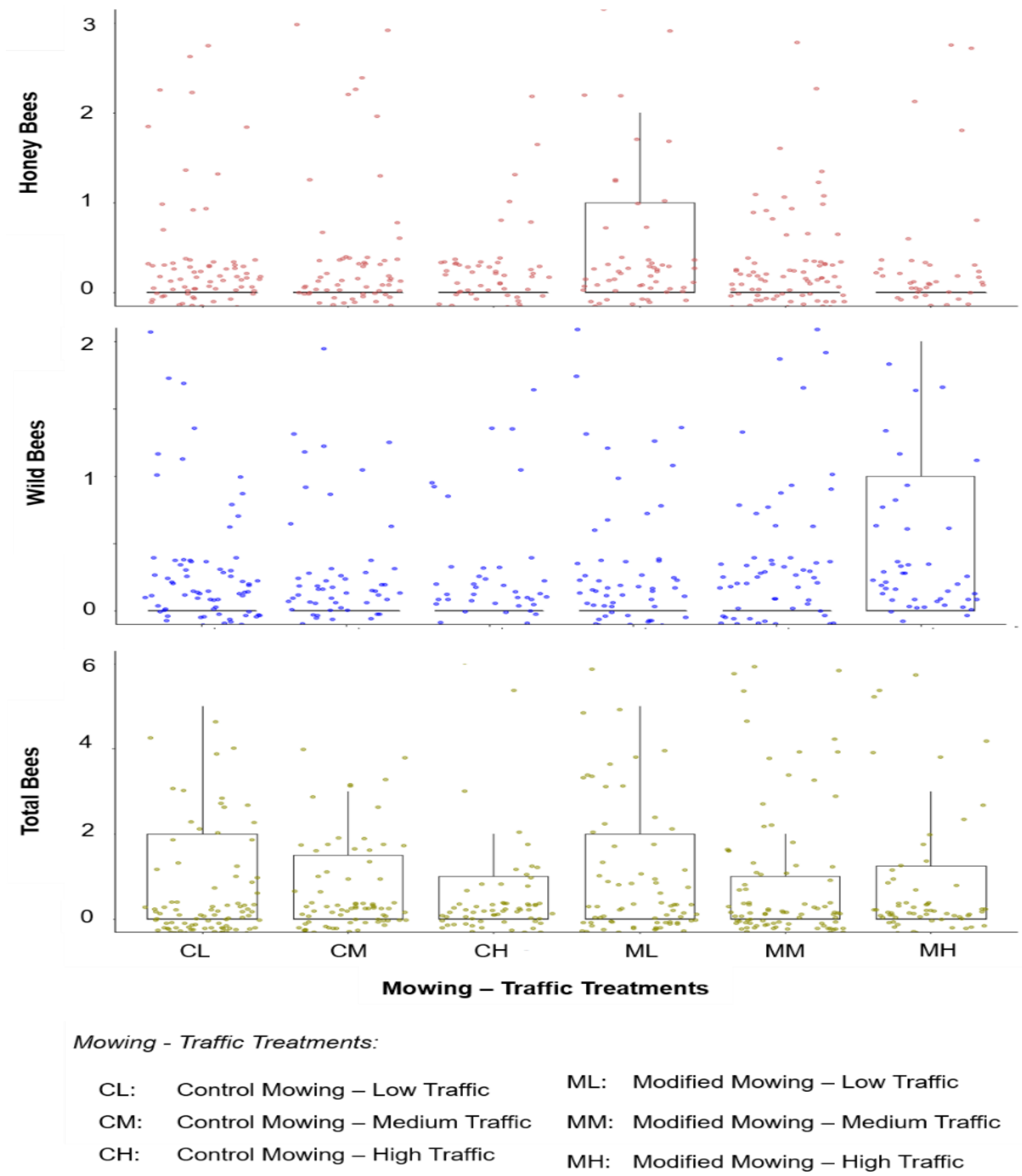


Figure 4: Boxplot comparisons of traffic-mowing treatment honey bees (top, red), wild bees (middle, blue), and total bees (bottom, yellow) using raw data. The black line within each of the boxes represents the median. The box represents the quartile range and the lines extending above and below the box represents the maximum and minimum values, respectively. Outliers were excluded from the visual. The circles represent individual observations. Using the estimated marginal means, there were no significant differences in honey, wild, or total bees observed across the mowing-traffic treatments.

I performed 316 scorecard assessments from 174 sampling locations during the 2020 field season. The average habitat quality rating was 33.67 (SD=14.66, n=316), placing it in the basic habitat quality category (scores between 21-35). The average habitat quality rating, when compared between all six treatments, was highest at the modified mowing - medium traffic sites with a mean rating of 36.18 (n=61, SD=14.24). This score places modified mowing - medium traffic sites, on average, to be in the next highest habitat quality ranking: moderate habitat quality (scores between 36-50). The mean width of the roadside ROW was 15.86 meters (SD=8.80). The width is inclusive of the ~5 meter safety strip that exists at the edge of the habitat.

I found habitat quality scores were not significantly associated with treatment or ROW width using analysis of variance (Table 4). I also found that visit had a significant impact on the results of the model (chi square coefficient = 20.63, $p<0.001$) (Table 4, Figure 5). The estimated marginal means found that the six mowing-traffic treatments did not have any significant differences in habitat quality rating. When the six treatments in visit 1 were compared to their visit 2 counterpart, there were significant differences present due to time.

Table 4: Analysis of variance results from the scorecard with the fixed effects width, visit, and the roadside mowing - road traffic treatments. I found visit to be significantly positively correlated with scorecard habitat quality rating.

	Chisq	Df	Pr(>Chisq)
ROW Width	2.13	1	0.14
Visit	20.63	1	<0.001
Mowing-Traffic Treatment	6.37	5	0.27

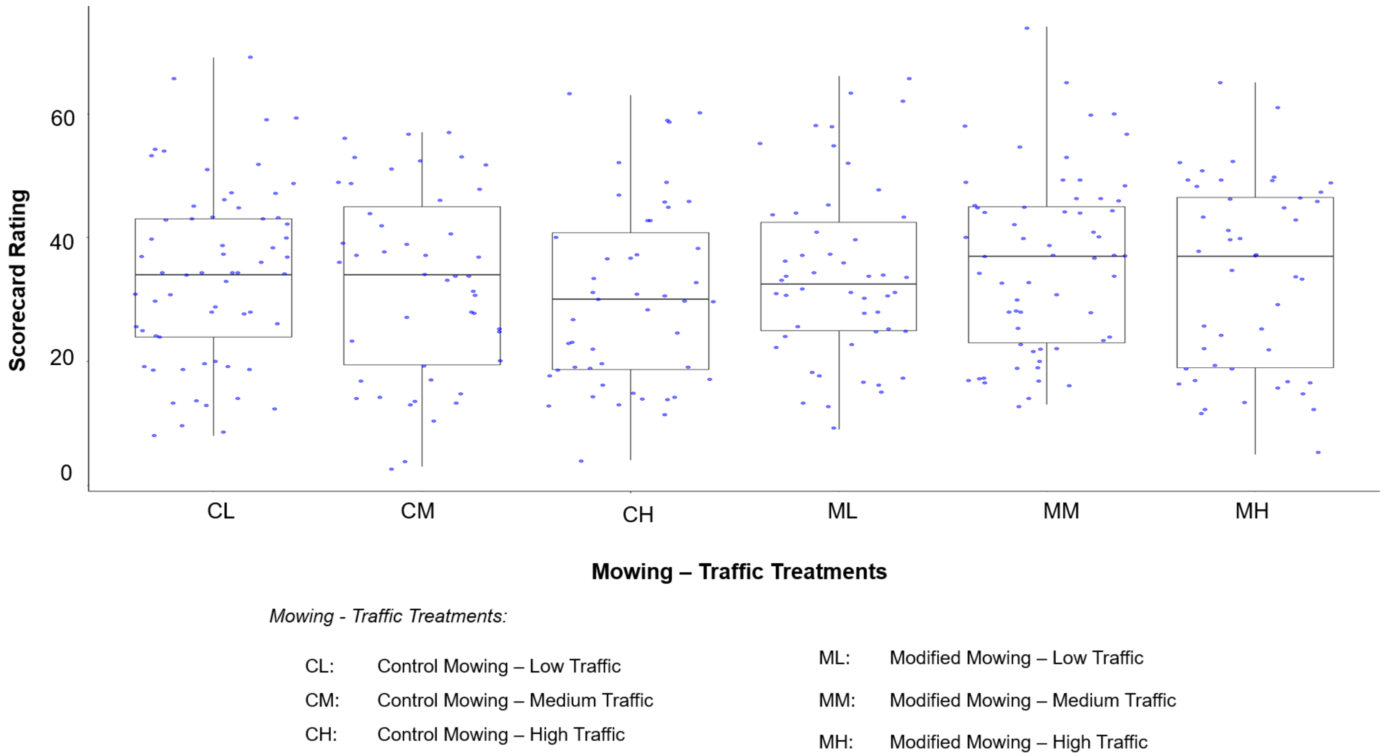


Figure 5: Boxplot comparison of traffic-mowing treatment for the scorecard habitat quality rating using raw data. The black line within each of the boxes represents the median. The box represents the quartile range and the lines extending above and below the box represents the maximum and minimum values, respectively. The blue circles represent individual observations. Using the estimated marginal means, there were no significant differences in scorecard rating observed across the mowing-traffic treatments.

Vegetation Survey

I performed 534 vegetation surveys from 177 sampling locations during 2019 (n=202) and 2020 (n=332). The average proportion of Asteraceae and Fabaceae cover during the first year was 16.21% (SD=23.34%) whereas the average cover was 21.51% (SD=25.34%). The control mowing - high traffic sites had the lowest proportion of these plants (mean=13.47%, SD= 19.43%) whereas the modified mowing - medium traffic sites had the highest proportion (mean=27.98%, SD=28.72%) 20.79% (n=111) of all observations (n=534) had no Asteraceae or Fabaceae in the transect.

Using analysis of variance, the proportion of Asteraceae and Fabaceae proportion in the transect was not significantly associated with treatment (Table 5, Figure 6). However, visit had a significant positive association on the proportion of Asteraceae and Fabaceae cover (chi square=4.69, p=0.03) (Table 5). The estimated marginal means found that the six mowing-traffic treatments did not have any significant differences in proportion of Asteraceae and Fabaceae cover. When the six treatments in 2019 were compared to their 2020 counterpart, there were no significant differences present due to time.

Table 5: Analysis of variance results from the vegetation proportion year and the roadside mowing - road traffic treatments as fixed effects. I found sampling year to be significantly positively correlated with the proportion of Asteraceae and Fabaceae observed.

	Chisq	Df	Pr(>Chisq)
Year	4.69	1	0.03
Mowing-Traffic Treatment	4.58	5	0.47

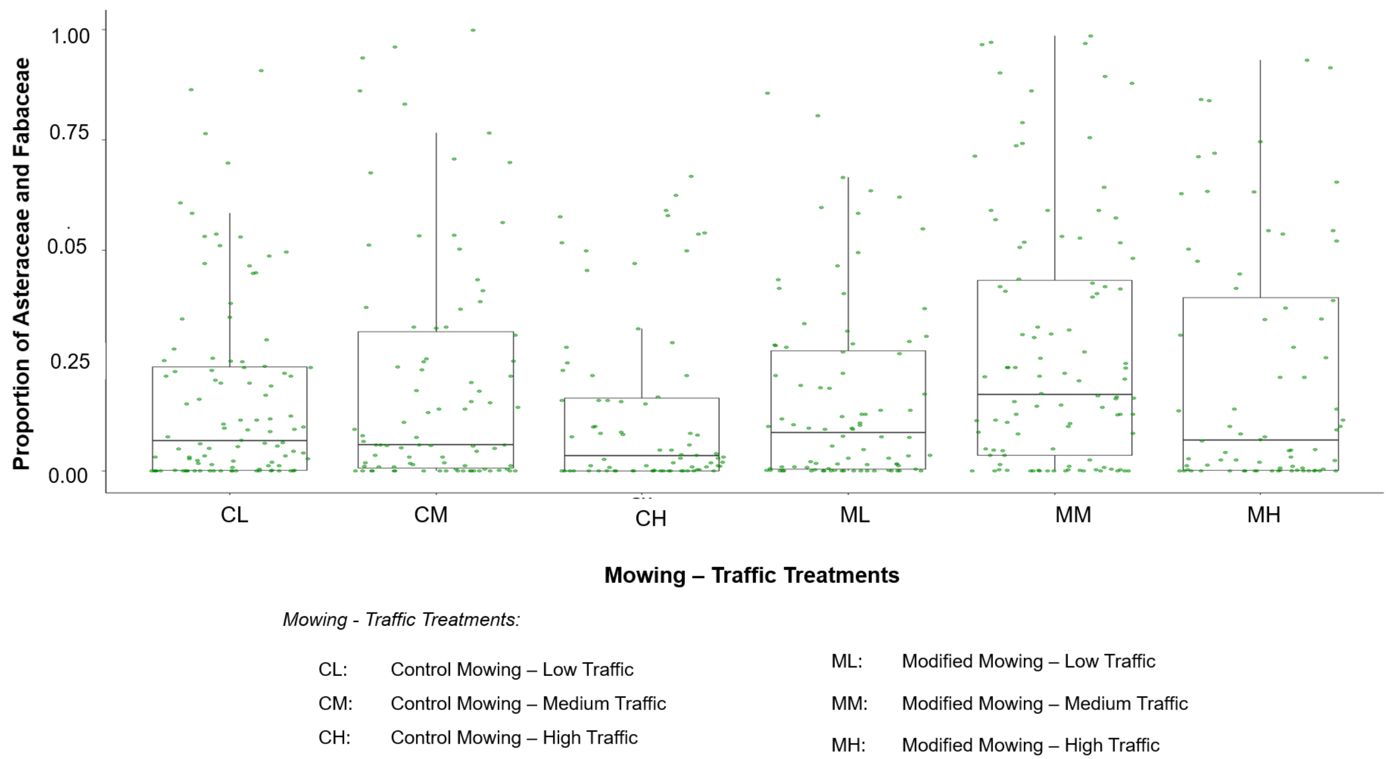


Figure 6: Boxplot comparison of traffic-mowing treatment for the proportion of Asteraceae and Fabaceae in the transect using raw data. The black line within each of the boxes represents the median. The box represents the quartile range and the lines extending above and below the box represents the maximum and minimum values, respectively. The green circles represent individual observations. Using the estimated marginal means, there were no significant differences in proportion of Asteraceae and Fabaceae observed across the mowing-traffic treatments.

Discussion

By analyzing different measures of habitat quality in roadside ROWs, I did not find evidence that the interaction between roadside mowing and road traffic may be contributing to variation in habitat quality. Of the three methods, there was not a significant difference in the estimated marginal means when examined as the six mowing - traffic treatments. However, visit and year were positively correlated with habitat quality measures. I found a significant difference in treatment when year and visit were added. Significant differences in treatment caused by time were present in the wild bee, honey bee, and scorecard observations. While the six treatments did not impact the variation of habitat quality, seasonal variability and interannual differences have the potential to influence habitat quality.

When measuring habitat quality using the scorecard, visit was positively correlated with this measure. The positive correlation between the habitat quality assessment and visit potentially supports the idea that plant phenology plays a role in the quality of habitat. Similarly, the estimated marginal means found significant differences between treatments and their later season counterpart. There were also significant differences between different treatments when visit was included. However, because there was no difference between the six treatments these differences may be due to seasonal variability. The importance of visit as a measure of seasonality may indicate that as the growing season goes on, more environmental conditions such as floral resources, temperature, and/or precipitation are present and within the preferred ranges of insect pollinators and plants. Butterflies have been observed to have higher abundance when mowing was reduced or shifted away from the period of peak butterfly activity (Halbritter et al., 2015). However, the peak activity of insect pollinators and the blooming period of floral resources differ across taxa and between species, so a universal time to mow the habitat is not possible. Understanding the plant and insect pollinator community composition in a given roadside environment will be needed in order to tailor mowing to benefit target insect pollinators.

When measuring habitat quality as wild bee abundance, honey bee abundance, and the proportion of Asteraceae and Fabaceae cover, year was positively correlated with these measures. When the wild bee, honey bee, and the proportion of Asteraceae and Fabaceae cover across the six treatments were compared using estimated marginal means, there was no significant difference found. There were significant differences between treatments when year was included. However, the estimated marginal means only found significant differences between treatments from 2019 and their 2020 counterpart in wild bee and honey bee observations. However, because there was no difference between the six treatments themselves, these differences may be due to annual variability. It is possible that because year is positively associated with the different habitat quality methods, indicating that studying altered mowing roadsides may need to be conducted over longer periods of time. Year has been found to influence changes in community function and community assemblage (Werner et al., 2020). This study took place over two field seasons. However, much of the literature that exists on roadside mowing is similarly constrained to one field season (Phillips et al., 2019; Saarinen et al., 2005), with the exception of the three-year study period studied by Noordijk et al. (2009).

It is possible that I did not find an effect of mowing because the study period was short, lasting for just two years. Short term studies are not able to account for potential long-lasting impacts of altered mowing treatment and the interannual and seasonal variations (Jeusset et al., 2016, Leston et al., 20). Russell et al. (2018) states that long term (40-50 years) vegetation management in powerline ROWs have the ability to support thriving bee communities.

Exploring the changes in roadside vegetation through reduced mowing over a longer period of time will be critical to understanding the true role of time, year and season, in habitat quality. It is possible that it will take several years before the full impacts of changes to habitat have on the plant and insect pollinator populations. Future research should emphasize the importance of long-term changes in roadside habitat quality and insect pollinator community assemblages. The scorecard provided a broad understanding of overall habitat quality and may be valuable to understanding long term trends in habitat quality in the presence, or absence of altered mowing management practices.

The lack of differences in habitat quality across treatments may be an indication that road traffic at all levels acts as a disturbance for pollinators and roadsides. This finding would be different from past research. Insect pollinators have been seen to have lower abundance in high traffic areas when compared to low traffic areas (Phillips et al., 2020; Skórka et al., 2013). However, I did not find this despite using a wide range of traffic volumes. The traffic in NYS encompassed a broad range of average vehicles per day on the stretches of road that I established sampling locations at. The maximum number of average vehicles per day in this study was 53,043. Past research on the response insect pollinators have to road traffic have primarily focused on roads with average daily traffic of less than ~10,000 cars (McCleery et al., 2015; Melis et al., 2010; Phillips et al., 2020; Saarinen et al., 2005; Skórka et al., 2013; Soluk et al., 2011). Munguira & Thomas (1992) and Soluk et al. (2011) exceptionally used roadside ROWs adjacent to roads with an excess of 25000 vehicles per day. These studies found that high traffic had no impact on ground beetle richness (Melis et al., 2010) nor butterfly abundance (Saarinen et al., 2005) or a negative impact on butterfly populations through increased on-road deaths (Skórka et al., 2013) but the high traffic in these studies were closer to the low and medium traffic levels that were established in my study. Similarly, to my results, research inclusive of higher traffic roads found that road traffic had no significant effect on butterfly (Munguira & Thomas, 1992) nor dragonfly populations (Soluk et al., 2011). It is possible that the negative impacts found in the high traffic found in the literature can be applied to the low and medium traffic levels in my study. This may mean that the high traffic areas in my study were worse for insect pollinators in such a way that other changes to the environment, such as through reduced mowing, may not have been able to overcome the disturbances that come with high traffic.

The high proportion of zeros found in the streamlined bee monitoring protocol data and the low proportion of Asteraceae and Fabaceae cover may further indicate that traffic at all levels contributes to roadside ROWs potentially not being suitable habitats for bees. Traffic has been understood to be a disturbance to plant growth, diversity, and abundance in roadsides (Wrzesień & Denisow, 2016). The inclusion of high traffic roads and roadsides in this study may explain the low proportion of Asteraceae and Fabaceae cover in roadsides. While Asteraceae and Fabaceae plants are established food sources for insect pollinators due to their high nectar and pollen content, they are not occupying a majority of roadside vegetation (Figure 5). These plant families are heavily utilized by bees for foraging, so the low proportion of Asteraceae and Fabaceae plants could help explain the high number of zeroes present in the streamlined bee monitoring protocol results. Williams et al. (2001) stated that Asteraceae may be able to predict the abundance and diversity of wild bees. Previous studies on the response of insect pollinators to reduced mowing have found unmowed/reduced mowing can yield higher abundance of butterflies (Saarinen et al., 2005) and higher flower species richness and pollinator abundance (Phillips et al., 2019). It is possible benefits from reduced mowing may not have been able to outperform the disturbances due to traffic, leading to no benefits or detriments to floral resources in roadsides. However, the multitude of confounding factors found in roadside environments makes it difficult to pinpoint specific motivators for habitat quality and insect pollinator presence.

The width of a roadside ROW did have an association on scorecard habitat quality rating. This differs from what has been observed in much of the research on roadside ROW habitats (Munguira & Thomas, 1992; Phillips et al., 2020; Saarinen et. al., 2005; Skórka et al., 2013). Monasterolo et al. (2020) found that wider ROWs that are intensively managed have higher plant species richness and pollinator abundance. The greater width can provide more area for plants to grow and increase in floral cover present. Hopwood (2008) studied roadsides at 14 different rural roads in Kansas and found that ROWs as narrow as 18 meters may provide quality habitat for bees. The width of the sampling locations found in my study ranged from 3 to 46.6 meters with an average of 15.86 meters (SD=8.80). My findings do not indicate that roadside width can provide quality habitat for bee or other insect pollinators. Hopwood's (2008) use of rural roads, that likely have low traffic, could be contributing to the difference between our results as I utilized high traffic roads. The possible differences in traffic could help account for the conflicting results between our studies, further indicating that traffic may be playing a large role in habitat quality. Like reduced roadside mowing, roadside ROW width may not be enough to overcome issues associated with high traffic.

Measuring roadside habitat quality using the streamlined bee monitoring protocol and the Asteraceae and Fabaceae cover are focused on the responses of bees. The scorecard focused on more habitat quality for insect pollinators generally. It is possible that other insect pollinators such as butterflies, beetles, and flies may benefit from roadside ROWs more than the wild and honey bees. Different taxa have already shown to respond differently to traffic and reduced mowing, so using bees and common bee forage plants as an indicator of habitat quality for all insect pollinators may not indicate the true potential of roadside habitats (Halbritter et al., 2015; Kasten et al., 2016; Leston et al., 2019; Noordijk et al., 2009; Phillips et al., 2019; Saarinen et al., 2005). The scorecard found that the average habitat quality rating was “basic”, which is defined as having limited, but some of the necessary components needed for insect pollinator habitat (Rights-of-way as Habitat Working Group, 2019). It recommends potential changes to vegetation management or revegetating the area (Rights-of-way as Habitat Working Group, 2019). It is possible that changes to mowing in higher traffic roadsides may be beneficial to other insect pollinators that just are not reflected through the tests used here.

Ultimately, I did not find evidence to date that reduced mowing over a 2-year period is associated with increased habitat quality for insect pollinators, across all traffic levels, in highways across upstate NYS. More research will be needed to understand which on-road and roadside conditions may be associated with higher habitat quality for insect pollinators. Although roadside ROWs can be heavily disturbed, they may prove to be critical complementary habitats to support insect pollinators enduring anthropogenic changes to the environment.

Bibliography

Ahrné, K., Bengtsson, J., & Elmqvist, T. (2009). Bumble Bees (Bombus spp) along a Gradient of Increasing Urbanization. PloS One, 4(5). https://doi.org/oi:10.1371/journal.pone.0005574

Bernes, C., Bullock, J. M., Jakobsson, S., Rundlöf, M., Verheyen, K., & Lindborg, R. (2016). How are biodiversity and dispersal of species affected by the management of roadsides? A systematic map protocol. Environmental Evidence, 5(1), 1–7. https://doi.org/10.1186/s13750-016-0055-x

Brooks, M. E., Kristensen, K., van Benthem, K. J., Magnusson, A., Berg, C. W., Nielsen, A., … Bolker, B. M. (2017). {glmmTMB} Balances Speed and Flexibility Among Packages for Zero-inflated Generalized Linear Mixed Modeling. The R Journal, 9(2), 378–400.

Buri, P., Humbert, J. Y., & Arlettaz, R. (2014). Promoting pollinating insects in intensive agricultural matrices: Field-scale experimental manipulation of hay-meadow mowing regimes and its effects on bees. PLoS ONE, 9(1), e85635. https://doi.org/10.1371/journal.pone.0085635

Cameron, S. A., Lozier, J. D., Strange, J. P., Koch, J. B., Cordes, N., Solter, L. F., & Griswold, T. L. (2011). Patterns of widespread decline in North American bumble bees. Proceedings of the National Academy of Sciences of the United States of America, 108(2), 662–667. https://doi.org/10.1073/pnas.1014743108

Carvell, A. C., Osborne, J. L., Bourke, A. F. G., Freeman, S. N., Pywell, R. F., Applications, E., & Carvell, C. (2015). Bumble bee species' responses to a targeted conservation measure depend on landscape context and habitat quality. Ecological Applications, 21(5), 1760–1771.

Colla, S. R. (2016). Status, Threats and Conservation Recommendations for Wild Bumble Bees (Bombus spp.) in Ontario, Canada: A Review for Policymakers and Practitioners. Natural Areas Journal, 36(4), 412–426. https://doi.org/10.3375/043.036.0408

Davis, A. K., Schroeder, H., Yeager, I., & Pearce, J. (2018). Effects of simulated highway noise on heart rates of larval monarch butterflies, danaus plexippus: Implications for roadside habitat suitability. Biology Letters, 14(5). https://doi.org/10.1098/rsbl.2018.0018

Esri Inc., (2021). ArcGIS Pro (Version 2.7.2). Esri Inc. https://www.esri.com/en-us/arcgis/products/arcgis-pro/.

Halbritter, D. A., Daniels, J. C., Whitaker, D. C., & Huang, L. (2015). Reducing Mowing Frequency Increases Floral Resource and Butterfly (Lepidoptera: Hesperioidea and Papilionoidea) Abundance in Managed Roadside Margins. Florida Entomologist, 98(4), 1081–1092. https://doi.org/10.1653/024.098.0412

Hopwood, J. L. (2008). The contribution of roadside grassland restorations to native bee conservation. Biological Conservation, 141(10), 2632–2640. https://doi.org/10.1016/j.biocon.2008.07.026

Hopwood, J., Black, S., & Fleury, S. (2015). Roadside Best Management Practices that Benefit Pollinators. 94. Retrieved from https://www.environment.fhwa.dot.gov/env_topics/ecosystems/Pollinators_Roadsides/BMPs_pollinators_landscapes_hi-rez.pdf

Jeusset, A., Vargac, M., Bertheau, Y., Coulon, A., Deniaud, N., Flamerie De Lachapelle, F., … Sordello, R. (2016). Can linear transportation infrastructure verges constitute a habitat and/or a corridor for biodiversity in temperate landscapes? A systematic review protocol. Environmental Evidence, 5(1), 1–11. https://doi.org/10.1186/s13750-016-0056-9

Kasten, K., Stenoien, C., Caldwell, W., & Oberhauser, K. S. (2016). Can roadside habitat lead monarchs on a route to recovery? Journal of Insect Conservation, 20(6), 1047–1057. https://doi.org/10.1007/s10841-016-9938-y

Keilsohn, W., Narango, D. L., & Tallamy, D. W. (2018). Roadside habitat impacts insect traffic mortality. Journal of Insect Conservation, 22(2), 183–188. https://doi.org/10.1007/s10841-018-0051-2

Kercher, S. M., Frieswyk, C. B., & Zedler, J. B. (2003). Effects of sampling teams and estimation methods on the assessment of plant cover. Journal of Vegetation Science, 14(6), 899–906. https://doi.org/10.1111/j.1654-1103.2003.tb02223.x

Khalid, N., Hussain, M., Young, H. S., Boyce, B., Aqeel, M., & Noman, A. (2018). Effects of road proximity on heavy metal concentrations in soils and common roadside plants in Southern California. Environmental Science and Pollution Research, 25(35), 35257–35265. https://doi.org/10.1007/s11356-018-3218-1

Lázaro-Lobo, A., & Ervin, G. N. (2019). A global examination on the differential impacts of roadsides on native vs. exotic and weedy plant species. Global Ecology and Conservation, 17. https://doi.org/10.1016/j.gecco.2019.e00555

Lenth, R. V. (2021). emmeans: Estimated Marginal Means, aka Least-Squares Means, https://cran.r-project.org/package=emmeans/

Leonard, R. J., McArthur, C., & Hochuli, D. F. (2018). Habitat complexity does not affect arthropod community composition in roadside greenspaces. Urban Forestry and Urban Greening, 30(January), 108–114. https://doi.org/10.1016/j.ufug.2018.01.016

Leston, L., & Koper, N. (2019). An urban wildlife habitat experiment: Conservation implications of altering management regimes on animals and plants along urban and rural rights-of-way. Journal of Urban Ecology, 5(1), 1–13. https://doi.org/10.1093/jue/juz013

Loffland, H. L., Polasik, J. S., Tingley, M. W., Elsey, E. A., Loffland, C., Lebuhn, G., & Siegel, R. B. (2017). Bumble bee use of post-fire chaparral in the central Sierra Nevada. Journal of Wildlife Management, 81(6), 1084–1097. https://doi.org/10.1002/jwmg.21280

Martin, A. E., Graham, S. L., Henry, M., Pervin, E., & Fahrig, L. (2018). Flying insect abundance declines with increasing road traffic. Insect Conservation and Diversity, 11(6), 608–613. https://doi.org/10.1111/icad.12300

McCleery, R. A., Holdorf, A. R., Hubbard, L. L., & Peer, B. D. (2015). Maximizing the wildlife conservation value of road right-of-ways in an agriculturally dominated landscape. PLoS ONE, 10(3), 1–19. https://doi.org/10.1371/journal.pone.0120375

McFrederick, Q. S., & LeBuhn, G. (2006). Are urban parks refuges for bumble bees Bombus spp. (Hymenoptera: Apidae)? Biological Conservation, 129(3), 372–382. https://doi.org/10.1016/j.biocon.2005.11.004

McNeil, D. J., McCormick, E., Heimann, A. C., Kammerer, M., Douglas, M. R., Goslee, S. C., … Hines, H. M. (2020). Bumble bees in landscapes with abundant floral resources have lower pathogen loads. Scientific Reports, 10(1), 1–12. https://doi.org/10.1038/s41598-020-78119-2

Melis, C., Olsen, C. B., Hyllvang, M., Gobbi, M., Stokke, B. G., & Røskaft, E. (2010). The effect of traffic intensity on ground beetle (Coleoptera: Carabidae) assemblages in central Sweden. Journal of Insect Conservation, 14(2), 159–168. https://doi.org/10.1007/s10841-009-9240-3

Munguira, M. L., & Thomas, J. A. (1992). Use of Road Verges by Butterfly and Burnet Populations, and the Effect of Roads on Adult Dispersal and Mortality. The Journal of Applied Ecology, 29(2), 316. https://doi.org/10.2307/2404501

Monasterolo, M., Poggio, S. L., Medan, D., & Devoto, M. (2020). Wider road verges sustain higher plant species richness and pollinator abundance in intensively managed agroecosystems. Agriculture, Ecosystems and Environment, 302(December 2019), 107084. https://doi.org/10.1016/j.agee.2020.107084

Muñoz, P. T., Torres, F. P., & Megías, A. G. (2015). Effects of roads on insects: a review. Biodiversity and Conservation, 24(3), 659–682. https://doi.org/10.1007/s10531-014-0831-2

New York State Department of Transportation. (2015). Annual Average Daily Traffic (AADT) Overview [PDF]. Retrieved from https://data.ny.gov/Transportation/Annual-Average-Daily-Traffic-AADT-Beginning-1977/6amx-2pbv

New York State Department of Transportation. (2017). Highway Design Manual [PDF]. Retrieved from https://www.dot.ny.gov/divisions/engineering/design/dqab/hdm/hdm-repository/chapt_14.pdf

New York State Department of Transportation. (2019). Traffic Data Viewer Geodatabase Shapefile [Dataset]. Retrieved from https://www.dot.ny.gov/divisions/engineering/applications/traffic-data-viewer/traffic-data-viewer-repository/NYSDOT_TDV_AADT_2019.gdb.zip

Nichols, R. N., Goulson, D., & Holland, J. M. (2019). The best wildflowers for wild bees. Journal of Insect Conservation, 23(5–6), 819–830. https://doi.org/10.1007/s10841-019-00180-8

Noordijk, J., Delille, K., Schaffers, A. P., & Sýkora, K. V. (2009). Optimizing grassland management for flower-visiting insects in roadside verges. Biological Conservation, 142(10), 2097–2103. https://doi.org/10.1016/j.biocon.2009.04.009

Ogilvie, J. E., Griffin, S. R., Gezon, Z. J., Inouye, B. D., Underwood, N., Inouye, D. W., & Irwin, R. E. (2017). Interannual bumble bee abundance is driven by indirect climate effects on floral resource phenology. Ecology Letters, 20(12), 1507–1515. https://doi.org/10.1111/ele.12854

Phillips, B. B., Gaston, K. J., Bullock, J. M., & Osborne, J. L. (2019). Road verges support pollinators in agricultural landscapes, but are diminished by heavy traffic and summer cutting. *Journal of Applied Ecology*, *56*(10), 2316–2327. https://doi.org/10.1111/1365-2664.13470

Phillips, B. B., Wallace, C., Roberts, B. R., Whitehouse, A. T., Gaston, K. J., Bullock, J. M., … Osborne, J. L. (2020). Enhancing road verges to aid pollinator conservation: A review. Biological Conservation, 250(July), 108687. https://doi.org/10.1016/j.biocon.2020.108687

R Core Team. (2020). R: A Language and Environment for Statistical Computing. Vienna, Austria. http://www.r-project.org

Ries, L., Debinski, D. M., & Wieland, M. L. (2001). Conservation value of roadside prairie restoration to butterfly communities. Conservation Biology, 15(2), 401–411. https://doi.org/10.1046/j.1523-1739.2001.015002401.x

Rights-of-way as Habitat Working Group. (2019). Rights-of-way as Habitat Working Group Pollinator Scorecard [PDF]. Retrieved from http://rightofway.erc.uic.edu/wp-content/uploads/Pollinator-Scorecard-Users-Guide-V2.1.pdf

Russell, K. N., Ikerd, H., & Droege, S. (2005). The potential conservation value of unmowed powerline strips for native bees. Biological Conservation, 124(1), 133–148. https://doi.org/10.1016/j.biocon.2005.01.022

Russell, K. N., Russell, G. J., Kaplan, K. L., Mian, S., & Kornbluth, S. (2018). Increasing the conservation value of powerline corridors for wild bees through vegetation management: an experimental approach. Biodiversity and Conservation, 27(10), 2541–2565. https://doi.org/10.1007/s10531-018-1552-8

Saarinen, K., Valtonen, A., Jantunen, J., & Saarnio, S. (2005). Butterflies and diurnal moths along road verges: Does road type affect diversity and abundance? Biological Conservation, 123(3), 403–412. https://doi.org/10.1016/J.BIOCON.2004.12.012

Skórka, P., Lenda, M., Moroń, D., Kalarus, K., & Tryjanowski, P. (2013). Factors affecting road mortality and the suitability of road verges for butterflies. Biological Conservation, 159, 148–157. https://doi.org/10.1016/j.biocon.2012.12.028

Soluk, D. A., Zercher, D. S., & Worthington, A. M. (2011). Influence of roadways on patterns of mortality and flight behavior of adult dragonflies near wetland areas. Biological Conservation, 144(5), 1638–1643. https://doi.org/10.1016/j.biocon.2011.02.015

Theodorou, P., Baltz, L. M., Paxton, R. J., & Soro, A. (2020). Urbanization is associated with shifts in bumblebee body size, with cascading effects on pollination. Evolutionary Applications, (July 2020), 53–68. https://doi.org/10.1111/eva.13087

Thomson, D. M. (2016). Local bumble bee decline linked to recovery of honey bees, drought effects on floral resources. Ecology Letters, 19(10), 1247–1255. https://doi.org/10.1111/ele.12659

U.S. Department of Transportation. (2019). Highway Finance Data Collection. Retrieved from Office of Highway Policy Information website: https://www.fhwa.dot.gov/policyinformation/pubs/hf/pl11028/chapter1.cfm

Vanbergen, A. J., Garratt, M. P., Vanbergen, A. J., Baude, M., Biesmeijer, J. C., Britton, N. F., … Wright, G. A. (2013). Threats to an ecosystem service: Pressures on pollinators. Frontiers in Ecology and the Environment, 11(5), 251–259. https://doi.org/10.1890/120126

Volenec, Z. M., & Dobson, A. P. (2019). Conservation value of small reserves. Conservation Biology, 0(0), 1–14. https://doi.org/10.1111/cobi.13308

Ward, K., D. Cariveau, E. May, M. Roswell, M. Vaughan, N. Williams, R. Winfree, R. Isaacs, and K. Gill. 2014. Streamlined Bee Monitoring Protocol for Assessing Pollinator Habitat. 16 pp. Portland, OR: The Xerces Society for Invertebrate Conservation.

Werner, C. M., Stuble, K. L., Groves, A. M., & Young, T. P. (2020). Year effects: Interannual variation as a driver of community assembly dynamics. Ecology, 101(9), 1–8. https://doi.org/10.1002/ecy.3104

Williams, N. M., Cariveau, D., Winfree, R., & Kremen, C. (2011). Bees in disturbed habitats use, but do not prefer, alien plants. Basic and Applied Ecology, 12(4), 332–341. https://doi.org/10.1016/j.baae.2010.11.008

Winfree, R., Aguilar, R., Vázquez, D. P., LeBuhn, G., & Aizen, M. A. (2009). A Meta-Analysis of Bees' Responses to Anthropogenic Disturbance. Ecology, 90(8), 2068–2076. from http://www.jstor.org/stable/25592723

Wojcik, V., & Buchmann, S. (2012). Pollinator Conservation And Management On Electrical Transmission And Roadside Rights - Of - Way : A Review. Journal of Polination Ecology, 7(3), 16–26.

Wood, T. J., Michez, D., Paxton, R. J., Drossart, M., Neumann, P., Gérard, M., … Vereecken, N. J. (2020). Managed honey bees as a radar for wild bee decline? Apidologie, 51(6), 1100–1116. https://doi.org/10.1007/s13592-020-00788-9

Wrzesien, M., & Denisow, B. (2016). Distribution and abundance of bee forage flora across an agricultural landscape - Railway embankments vs. road verges. Acta Societatis Botanicorum Poloniae, 85(3), 1–15. https://doi.org/10.5586/asbp.3509

Chapter 2: Bumble Bee (*Bombus spp.*) abundance not impacted by roadside mowing and road traffic in highway roadsides across upstate New York

Abstract

Many pollinating insects, including native pollinators such as bumble bees (*Bombus spp.*), are facing population declines globally due to habitat loss and other anthropogenic factors. The grassy areas next to roads, known as roadside rights-of-way (ROWs), are potential alternative habitats for bumble bees. However, roadside ROWs are highly disturbed areas, due roadside management practices such as mowing and on-road traffic. My research objective was thus to investigate if and how road traffic level and roadside mowing interact to influence bumble bee abundance in highway roadside ROWs across New York State. I tested this using a sweep netting and photography in 2019 and 2020, along 30 highways with Control Mowing- Low Traffic (n=33 sampling locations), Control Mowing - Medium Traffic (n=29), Control Mowing - High Traffic (n=27), Modified Mowing - Low Traffic (n=28), Modified Mowing - Medium Traffic (n=35), and Modified Mowing - High Traffic (n=25) treatments. I was unable to test the effect of mowing and traffic treatment on bumble bee abundance, due to the extremely limited number of bumble bees observed (98% of n=916 observations across two methods and two years found 0 *Bombus spp.*). This was potentially due to insufficient foraging plant species and/or areas for nesting, components critical for bumble bee survival and habitat. Further research is needed to know if and how roadside ROWs can support bumble bees.

Introduction

Pollinating insects such as bumble bees (*Bombus spp.*) are critical providers of ecosystem services, providing pollination of plants and biodiversity by maintaining the diversity of plant life. They are critical to the pollination of native flowers and crops in temperate climates (Cameron et al., 2011). The services provided by bumble bees may be at risk because many pollinating insect populations, including wild bumble bees are decreasing worldwide (Cameron et al., 2011, Hopwood, 2008; Vanbergen et al., 2013).

However, bumble bees are considered to be declining and at risk of endangerment. While there is a lack of data to fully demonstrate the decline and the rate at which it is occurring, especially among different species, there is a growing body of research on this topic (Cameron et al., 2011; Colla, 2016; Richardson et al., 2019; Thomson 2016; Williams & Osborne, 2009). Direct threats to bumble bees include disease and competition with other pollinators (Colla, 2016; Thomson, 2016; Williams & Osborne, 2009). Additionally, native bumble bees are frequently outcompeted by other pollinators. The presence of honey bees may lead to the absence of bumble bees as well as smaller body sizes on average (Colla, 2016; Thomson, 2016). Honey bees are a greater advantage because of their large colony sizes and generalist foraging behaviors (Colla, 2016; Thomson, 2016). Indirectly, bumble bees and their conservation status are impacted by issues that affect plants. Bumble bees rely on the presence of suitable foraging plant species and areas for nesting, so degradation or removal of these resources can impact the suitability of the habitat and the survival of bumble bees. Plants are further put in adverse conditions through the application of pesticides and herbicides. Herbicides can further decrease food and habitat while potentially having sublethal impacts bumble bee populations (Colla, 2016; MacPhail et al., 2019: Williams & Osborne, 2009). These changes, when combined with the outright removal of natural habitat for human development leaves limited available habitat for bumble bee populations. The loss of floral resources has been linked with decreased abundance of bumble bees (Hopwood, 2008; Nichols et al., 2019; Ogilvie et al., 2017; Thomson, 2016). Detrimental changes caused by climate change, application of herbicides, and loss or degradation of habitat can, as a result, have adverse consequences on bumble bees (Colla, 2016; Thomson, 2016; Williams & Osborne, 2009).

Bumble bees are also impacted by anthropogenic factors, directly and indirectly. Human growth and development have contributed to changing environments carried out by intensive agricultural practices, urban development, and climate change. The effects caused by the removal of natural areas, and their floral resources, that bumble bees typically habituate are further exacerbated by climate change. Climate change is contributing to the shift in phenological and geographical ranges of bumble bees (Bartomeus et al., 2011; Colla, 2016; Ogilvie et al., 2017; Pyke et al., 2013). Shifts are also occurring in forage plant species that bumble bees rely on; however, the shifts may not be in the same direction or at the same rate (Bartomeus et al., 2011; Colla, 2016). Bumble bees have a mutualistic relationship with many plant species due to their ability to buzz-pollinate, releasing pollen from flowers by buzzing at a high frequency (Cameron et al., 2011; Colla, 2016). These differing ranges can result in decreased pollination and foraging, harming both foraging plant species and bumble bees.

The variety of threats to bumble bees have led to interest in understanding these threats further in order to better understand the potential declines associated with their populations. Because one of the largest threats to bumble bees is loss and degradation of habitat due to their reliance on foraging and nesting resources, researchers have begun exploring whether they can thrive in developed environments (Ahrné et al., 2009; McFrederick & LeBuhn, 2006; Winfree et al., 2009). It is unlikely that human development will slow down or cease, so understanding if developed environments have any potential as habitats are becoming increasingly important. Limited studies have been conducted on bumble bees in urban environments as the focus is primarily on agricultural settings. However, of the urban studies that exist, many examine bumble bees in urban greenspaces such as city parks and gardens (Ahrné et al., 2009; McFrederick & LeBuhn, 2006; Winfree et al., 2009).

There is mixed evidence that bumble bees can use these semi-natural habitats. In these urban environments, such as in city parks, bumble bees have continued to demonstrate the need for sufficient foraging plant species as well as areas for nesting in these areas (Ahrné et al., 2009; McFrederick & LeBuhn, 2006; Winfree et al., 2009). Bumble bees were observed to utilize plants in the parks and gardens, but these environments are disturbed and do not always have ideal habitat conditions. The management practices associated with parks and gardens, in order to emphasize aesthetics and safety, can remove substrate, such as brush piles and fallen trees, which are needed for nesting (McFrederick & LeBuhn, 2006). However, nesting is still possible as some bumble bees have been observed to use holes created by rodents as nest sites (McFrederick & LeBuhn, 2006). Management decisions and disturbance influence the presence of invasive and ornamental plant species in these areas, which are outside of the wild plants and crops that bumble bees are typically surrounded by (McFrederick & LeBuhn, 2006; Winfree et al., 2009). Although the resources needed for survival are present and can be considered potential alternative habitats, anthropogenic disturbances that are negatively associated with bumble bee abundance, richness, and/or diversity make them not ideal habitats (Ahrné et al., 2009; McFrederick & LeBuhn, 2006; Winfree et al., 2009).

Other potential developed areas that are being considered include rights-of-ways (ROWs). ROWs are linear corridors created for transportation and are accompanied by natural to semi-natural areas. ROWs are areas created for railways, utility lines, and roads. The areas next to the infrastructure are often vegetated, providing habitat and resources that are otherwise missing from surrounding developed landscapes (Gardiner et al. 2018; Villemey et al., 2018). These areas are often actively managed to support the infrastructure they are next to or under and can vary in composition and size.

Roadside ROWs are increasingly present in today's urban environment: The United States has the largest roadside network, covering 6,506,204 kilometers of road and 3,903,722 hectares for potential ROW habitat (Wojcik & Buchmann 2012).

Their ubiquitous nature offers an abundance of land for potential habitat on a large spatial level. Researchers are exploring how these areas are performing as habitat for plants, insects, and other organisms. Their widespread nature opens them to varying conditions. Variations in the plant community, traffic volume, and a variety of other issues present barriers to increasing insect pollinator abundance (Muñoz et al., 2015; Volenec & Dobson, 2019).

Roadside ROWs are often degraded and highly disturbed habitats, potentially impacting their ability to be a habitat for insect pollinators. Roadside habitat is impacted by the activity that is occurring on the road itself. Higher traffic roads are associated with higher disturbances because they are subject to more noise pollution, chemical deposition, collision with vehicles, and impediment of movement. Although there are no studies that examine the response of bumble bees to noise pollution, noise pollution has been shown to desensitize Monarch butterflies to stressors (Davis et al., 2018). The desensitization could potentially increase the risk of danger, especially when passing through roadside corridors (Davis et. al., 2018). It isn't clear how bumble bees respond to noise pollution, but it is possible that they would similarly. Invasive plant species using these corridors and chemical deposition of heavy metals and salt could interfere with the growth of native plant species that pollinators are dependent on (Khalid et al., 2018; Muñoz et al., 2015). Traffic may increase rates of mortality of traveling insects (Halbritter et al., 2015; Phillips et al., 2019). Collisions with vehicles may increase rates of mortality of animals and insects that live in ROWs (Keilsohn et al., 2018; Skórka et al., 2013; Soluk et al., 2011). Roads can act as a barrier to dispersal for pollinating insects (Mungaria & Thomas, 1992; Muñoz et al., 2015).

Despite the disturbances of roads, roadsides may be usable habitats for bumble bees. Bumble bees have the ability to forage within 2 km of their nesting site, indicating that traveling between patches of habitat is not necessarily a barrier to survival (McFrederick & LeBuhn, 2006; McNiel et al., 2020; Theodorou et al., 2020). Although bumble bees are able to travel to more distant floral resources, adequate foraging areas, with suitable floral resources, within this distance are critical. However, this may lead to restrictions in the areas in which nests are established (Russell et al., 2018) as well as lead to bumble bees spending more time foraging the resources needed for their colony (Pyke et al., 2011). This emphasizes the need for adequate habitat for pollinators as they cannot easily move to distant habitats. These factors can further diminish the quality of habitat and the abundance of pollinators that are able to live in these areas.

There is not a conclusive understanding of how road traffic impacts abundance of insect pollinators. Increased traffic has been observed to have no impact on the abundance of butterflies (Munguira & Thomas, 1992; Saarinen et al., 2005), decreased abundance of pollinators (Phillips et al., 2019), and increased abundance of butterflies killed (Skórka et al., 2013). This may be due to the road types used in these studies, primarily examining the effects of traffic on insect pollinators in ROWs having lower traffic densities (Phillips et al., 2020; Saarinen et al., 2005; Skórka et al., 2013). Studies that focus on urban, local, and rural roads also do not typically include the ROWs adjacent to large highways, which are typically characterized by having high traffic (Munguira & Thomas, 1992; Phillips et al. 2020; Ries et al., 2001; Skórka et al., 2013, Skórka et al., 2018). Research that is inclusive of many traffic levels, which include higher traffic levels, is essential to understanding the variability of insect pollinators and which type of habitats should receive the most conservation effort.

Roadside ROWs are disturbed further by conditions that exist within the roadsides themselves. These include invasive species, herbicide and pesticide use, and intensive mowing and management practices (McCleery et al., 2015; Muñoz et al., 2015). Invasive plants often outcompete the beneficial native wildflowers that are best suited for bees, decreasing the preferred foraging plants (Williams et al., 2011; Wrzesień & Denisow, 2016). Management practices including the application of herbicides and pesticides and mowing have the ability to alter the composition and number of floral resources present for pollinators (Hopwood et al., 2015; Noordijk et al., 2009). Disturbances to floral resources can uniquely impact insect pollinators and their habitat.

The disturbance and removal of floral resources through mowing, a common management technique, used by managers of roadside ROWs, has the ability to impact insect pollinator foraging, habitat, and abundance. This management technique decreases the height of vegetation present for safety and aesthetic reasons. However, some mowing is needed for motorist visibility and to create space for cars to regain control and prevent collisions. The interior of ROWs are not necessarily subject to the same management practices as the edges. A variety of insect pollinators have been observed utilizing mowed roadsides environments as habitats (Halbritter et al., 2015; Keilsohn et al., 2018; Phillips et al., 2019; Noordijk et al., 2009; Ries et al., 2001; Saarinen et al., 2005; Skórka et al., 2013). Wild bees, including bumble bees, have been observed to increase in abundance in managed meadows and powerline corridors that are not mowed (Buri et al., 2014; Russell et al., 2018). These areas are different in their disturbances but are still managed in a similar fashion to roadside environments. Mowing in excess of 80% of the habitat, especially during the growing season, removes vital food sources and habitats for wild bees (Buri et al., 2014). However, there are no studies pertaining specifically to bumble bees and mowing. These studies do not look at bumble bees specifically but include them in the grouping of wild bees. Reduced mowing and mowing later during the growing season have been seen to increase pollinator abundance in some highway ROWs, likely due to the increased diversity in bloom timing through the shift in the groups of plants that are removed (Noordijk et al., 2009). Furthermore, roadside habitats that had been mowed exhibited fewer floral resources and pollinators (Phillips et al., 2019). However, Phillips et al. (2019) notes that results from short term studies may not fully display the response of pollinators to changes in management.

The results from changes made to ROWs vary across spatial and temporal scales. The majority of studies examining reduced mowing has been conducted over short study periods and small regions or sections of highways. Limited study size and length may not reveal how the changes in mowing patterns impact pollinators, leading to inconclusive or conflicting results. Existing conclusions on the response of insect pollinators to mowing reflect small spatial scales, many of which only examine one growing season, extending across a small number and size of sites. These sites are also within a single region or city, limiting the understanding of how abundance and mowing practices differ across diverse landscapes. Variations in locations and over several years must be accounted for in order to more completely understand if there is a benefit to reducing mowing. Location-specific climatic variations may also cause mowing to be more beneficial in one area than another, indicating the requirement for larger study size.

Roadside mowing and on road traffic pose some of the greatest disturbances to roadside environments, but if roadsides are to be considered alternative habitats where bumble bees can thrive, the interaction between the two needs to be fully understood. Disturbances to roadside environments are strongly tied to on road activities, both influencing the suitability of roadsides as habitats. Pollutants from vehicles end up in roadside habitats and are deposited into the soil which can impact the plants upon which bumble bees and other pollinators are reliant. A similar, potentially harmful, interaction between roadside mowing and on road traffic may work together to diminish resources for and populations of bumble bees in roadsides. The current understanding of roadside mowing and traffic exist independently of each other but do not assess how they interact and if this interaction has distinct effects on bumble bees and other insect pollinators.

In order to address if roadside habitats are suitable habitats for bumble bees, I studied the impacts of road traffic and roadside mowing on bumble bee abundance. Specifically, I assessed the associations between mowing practices in New York State (NYS) highway roadsides and traffic volume on the highways on bumble bees. I hypothesized that sites with low traffic and reduced mowing would have more bumble bees. I expected this because the abundance of wild bees and butterflies have been seen to increase in unmowed ROWs (Halbritter et al., 2015; Phillips et al., 2019; Russell et al., 2018). Bee abundance will likely also be higher in areas with low traffic because they have been observed to have increased pollinator abundance (Phillips et al., 2019) and richness of bee forage plants (Wrzesień & Denisow, 2016).

Materials and Methods

Study Sites

30 stretches of highway were selected across upstate New York (Figure 1). The stretches of highway were selected in collaboration with the NYSDOT to include a wide variety of road traffic, road size, speed limit, and surrounding land use that represents the diversity of roads and roadsides across the state. The study sites were established in spring 2019 and were maintained by the NYSDOT through fall 2020.

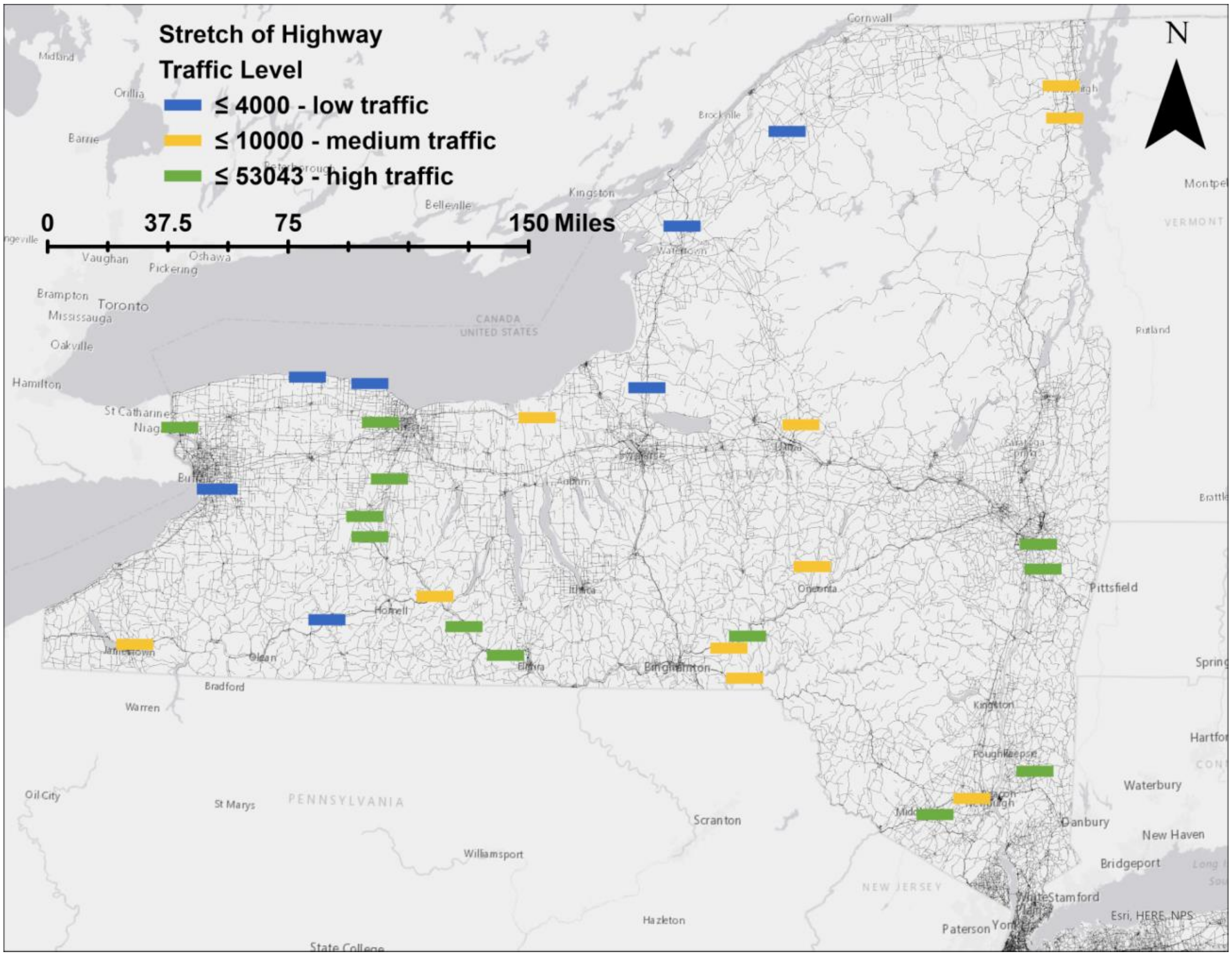


Figure 1: This map contains the average daily traffic at each of the sampling replicates (n=177) across NY. The blue dots represent sampling replicates that have average daily traffic less than 4,000 vehicles per day (low traffic, n=61). The yellow dots represent sampling replicates that have an average daily traffic range of 4,001 to 10,000 vehicles per day (medium traffic, n=64). The green dots represent sampling replicates that range from 10,001 to the maximum, 53,043 vehicles per day (high traffic, n=52). The NYS road network contains all of the roads present in NYS.

Treatments

At each site, there were two miles of each treatment (control and reduced mowing), with three sampling replicates within each treatment, spaced at least half a mile apart (Figures 2 & 3). The control treatment had the current NYSDOT mowing management practices applied to one section/side of the ROW (NYSDOT Vegetation Mowing Policy TMI 14-01). Under this mowing plan, it was standard for all interstates and primary highways to be mowed a single pass twice a year. Secondary highways were mowed a single pass once a year (New York State Department of Transportation, 2017).

Figure 2: A visual representation of the treatment design at each site. The orange strip on both sides of the road indicates the ~5m safety strip that is required to be mowed by regulation. The blue represents the modified mowing pattern that is wider and mowed less frequently than the control mowing pattern, as seen in green.

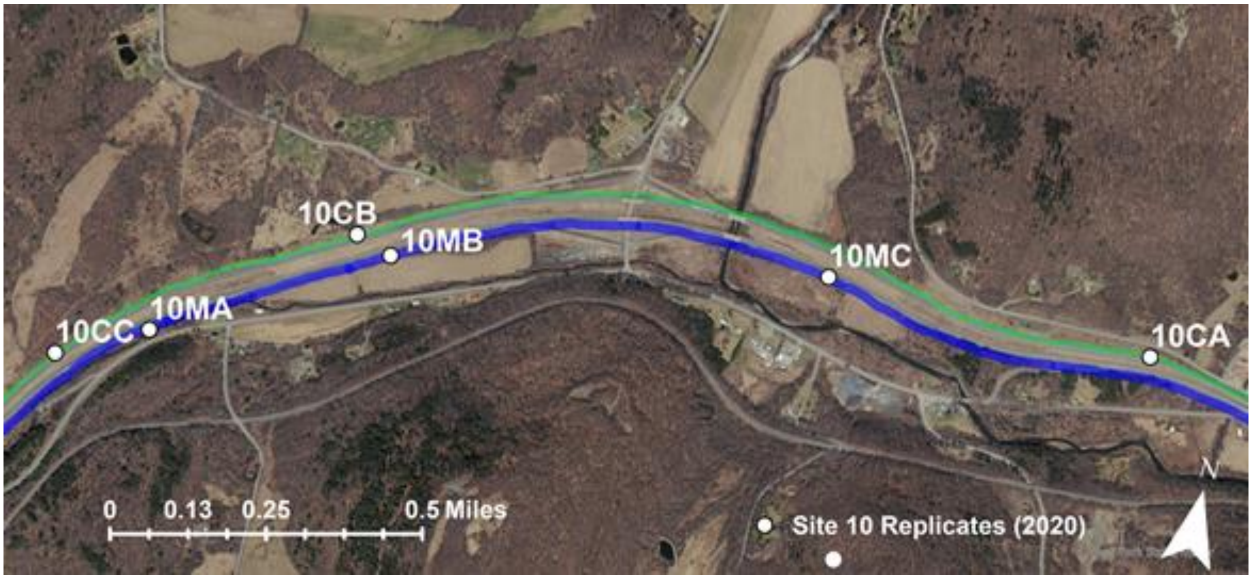


Figure 3: A diagram displaying the treatments and their three replicates for site 10, one of the 30 stretches of highway in the study. Each replicate was spread at least half a mile apart and was contained within the two-mile range allotted for the treatment (control or reduced mowing). The blue represents the modified mowing that is wider and mowed less frequently than the control mowing, as seen in green.

The other section/side of the highway, receiving the modified treatment was mowed every two years after plant-killing frost conditions (Figure 2). This mow was also a wider pass of the mower than exhibited in the control mowing conditions. These treatments were applied just beyond the safety strip. The NYSDOT was responsible for maintaining the treatments and sites for the two-year study period. The modified sites had no mowing the first year of the study (2019). In the second year (2020), modified sites were mowed later and wider.

The safety strip of the ROW is required to be mowed for motorist visibility, by NYS law. The New York State Department of Transportation requires that the first 15 feet of the ROWs are mowed and maintained as short, grassy vegetation in order to create a 'safety strip' (New York State Department of Transportation, 2017). This area provides a buffer to passing vehicles that may lose control of their vehicle (Figure 3).

Traffic

I obtained data on the road network and the counts of annual average daily traffic (AADT) from the NYS Department of Transportation via the NYS GIS Clearinghouse (New York State Department of Transportation, 2019). NYSDOT uses a variety of methods to determine the average daily traffic for each year. Counts were done through short counts using portable traffic counters, continuous counts from the Statewide Monitoring System and the Weigh-in-Motion (WIM) Stations that were positioned across the state (New York State Department of Transportation, 2015). I used the most recent available data. From my research thus far, the most recent AADT data was from the 2019 count statistics (New York State Department of Transportation, 2019).

I used the AADT data to assign traffic levels to each sampling location. AADT values from 2019 ranged from 0 to ~300,000 vehicles per day across NYS. I joined the 2019 roadway shape file (New York State Department of Transportation, 2019) with the 2019 count statistics table in ArcGIS Pro 2.7.2 (Esri Inc., 2021). The AADT values and the associated roads were overlaid with the coordinates of the sampling locations. I then classified the sampling locations by their AADT value (Figure 1). I considered a road to be low traffic if there were 4,000 or fewer vehicles per day, medium traffic if there were between 4,001 and 10,000 vehicles per day, and a high traffic road if there were between 10,001 and the maximum, 53,043 vehicles per day (Figure 1). Of the 177 sampling locations, the mean of the average number of vehicles per day was 9,268 (SD=11,106). These traffic levels were selected so that comparisons could be drawn between my results and the findings in the literature. The traffic ranges found in the literature are primarily along roads that are less than ~10,000 vehicles. I set up my low and medium traffic levels so that they would correspond with many of the intermediate and high levels found in the literature.

I then categorized my sampling locations (n=177) into six treatments representing interactions of on-road traffic level and roadside mowing treatment (Table 1). The six treatments included Control Mowing- Low Traffic (n=33 sampling locations), Control Mowing - Medium Traffic (n=29), Control Mowing - High Traffic (n=27), Modified Mowing - Low Traffic (n=28), Modified Mowing - Medium Traffic (n=35), and Modified Mowing - High Traffic (n=25) treatments.

Table 1: Summary table of the number of sampling replicates (n=177) within each treatment and traffic condition pairing (using the same traffic groupings as seen in Figure 1) for 2019 (n=161) and 2020 (n=174). Numbers vary between years due to slightly differing sampling locations used when reestablishing sites between years and because some replicates were not set up in one year but were in set up in the other.

Count of Each Treatment				
	2019		2020	
	Control	Modified	Control	Modified
Low	28	28	33	27
Medium	28	32	29	34
High	24	21	26	25

Bumble Bee Abundance Surveys

Sweep Netting

In both 2019 and 2020, I conducted standardized sweep netting to broadly survey insects. I used a canvas sweep net along the 100-meter transect and collected insects (Popic et al., 2013). This was performed twice per field season at each sampling location. These samples were placed in clear plastic bags and then in a cooler. I brought the samples back to the lab and frozen them for later identification to genus. Sweep netting in combination with the vegetation samplings allowed for an understanding of which pollinators were present along the vegetation transect as well as which plants were being utilized by bumble bees (Grundel et al., 2011).

Photography

To determine bumble bee abundance, in 2020, I visually observed foraging bumble bees that were present along a transect (50 m) in the middle of the ROW beyond the safety strip (Phillips et al., 2019). I walked in both directions of the transect at a steady pace for approximately 10 minutes. This method was adapted from a study that examined the abundance of multiple insect pollinator taxa, including bees, in a variety of locations, one of which being the center of roadsides (Phillips et al., 2019). In my study, I only observed the presence of bumble bees in the middle of the ROW (corresponding to their VC location, due to regulatory and landscape differences between the UK and US).

Surveys at each sampling location took place at least twice during the 2020 field season. The visits were separated by approximately 4-6 weeks in order to obtain data on the multiple blooming periods and *Bombus spp*. life cycles (Pyke et al. 2011). The field season began in May and continued through August, or until all sites were visited twice (Loffland et al., 2017). I conducted surveys when environmental conditions were suitable for bumble bee activity (between, between the hours of 10:00 A.M. and 5:30 P.M, and when ambient temperatures are above 13° Celsius, with partly cloudy skies, (40% cloud cover/you can still see your shadow) and wind speed less than 8 mph, or above 17° Celsius with any sky conditions) (Colla, 2016; Ward et al., 2014).

I photographed bumble bees within 1 m of either side of the transect and 2 m ahead and later identified them to species. If I noticed a bumble bee, it was photographed using the iPhone X camera (Richardson et al., 2019). The plant that the bumble bee was photographed on was recorded to species, or if the bee was in flight, which was noted instead (Cole et al., 2020; Loffland et al., 2017). Bumble bee were identified to species using the *Guide to Bumble Bees of the Eastern United States* (Colla et al., 2011). This guide provided example images, phenological and geographical ranges, and descriptions for each species of bumble bee in the eastern United States as well as a dichotomous key that helped distinguish between species.

Using photography as a means of identifying bumble bees to species has been recently growing in popularity, especially in the field of citizen science (MacPhail et al., 2019). In previous studies that include both citizen scientists and experienced scientists, photos were correctly identified to species between 68-95% of the time (Richardson et al., 2019; Suzuki-Ohno et al., 2017).

Statistical Analyses

I analyzed my data using R (R Core Team, 2020). My response variables were bumble bees photographed/caught. I included site, temperature, and cloud cover as independent variables in my analyses, in addition to the six treatment levels. These factors were included because bumble bee activity is highly dependent on weather conditions (Ahrné et al., 2009; Nichols et al., 2016; Theodorou et al., 2020; Thomson, 2016).

Results and Discussion

Results

Sweep Netting

I collected 584 sweep samples from the 177 sampling locations across 2019 and 2020 (2019 n=357, 2020 n=227). Of the total, 97.60% (n=584) had zero bumble bees. Only 14 of the total sweep samples included one or more bumble bees (Figure 4). The maximum number of bumble bees seen at a given site was 2 bees, which occurred at two sampling locations. The total sum of bumble bees caught using the sweep net was 16. Only one bumble bee was caught in 2020, all others were caught in 2019. 10 observations were at modified mowing sites and 4 observations at control mowing sites. Because of the large amount of samples that contained no bumble bees, I was unable to run the intended statistical analyses.

Photography

Across the 2020 samples (n=332), I was able to take pictures of 10 different bumble bees (Figure 5, Supplemental File 1). Nine of the 10 bumble bees photographed could be identified to species using the photos taken. One of the bees did not have enough pictures of a high enough quality, so it could not be identified. All nine of the bees that were identified were common eastern bumble bees (*Bombus impatiens*). Six bees were observed in flight or briefly landing before exiting the transect area. These bees were not identified to species but were counted. Bumble bees were found on plant families including Asteraceae, Fabaceae, Caprifoliaceae, Caryophyllaceae, Hypericaceae, Lamiaceae, and Lythraceae (Table 2).

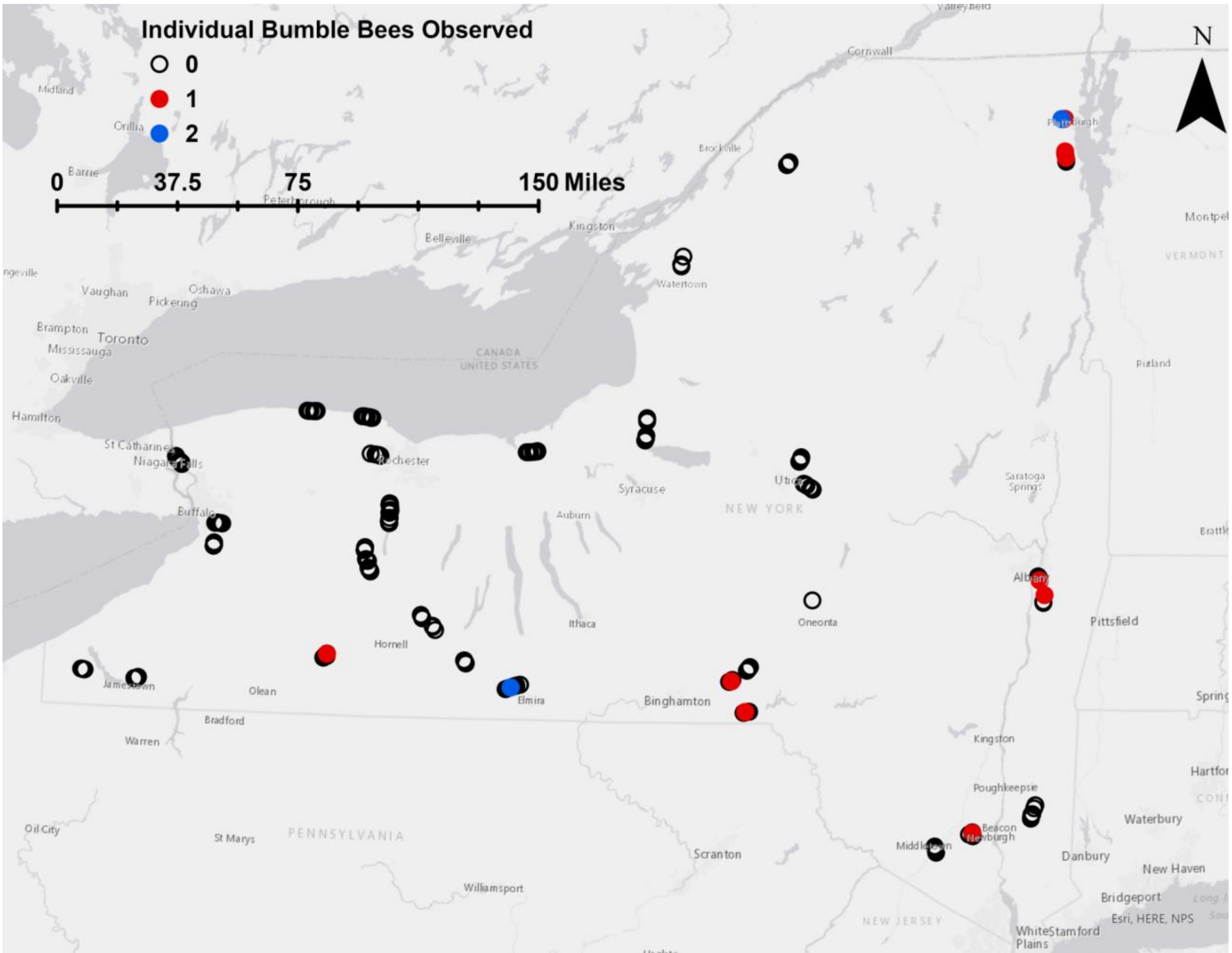


Figure 4: A map with the number of individual bumble bees caught during each sweep net observation (n=584). Sites where no bumble bees were found are shown as an outline of a circle. Sites where one bumble bee was observed are shown in red and sites where two were observed are shown in blue. Of the 584 observations, only 14 caught one or more bumble bees.

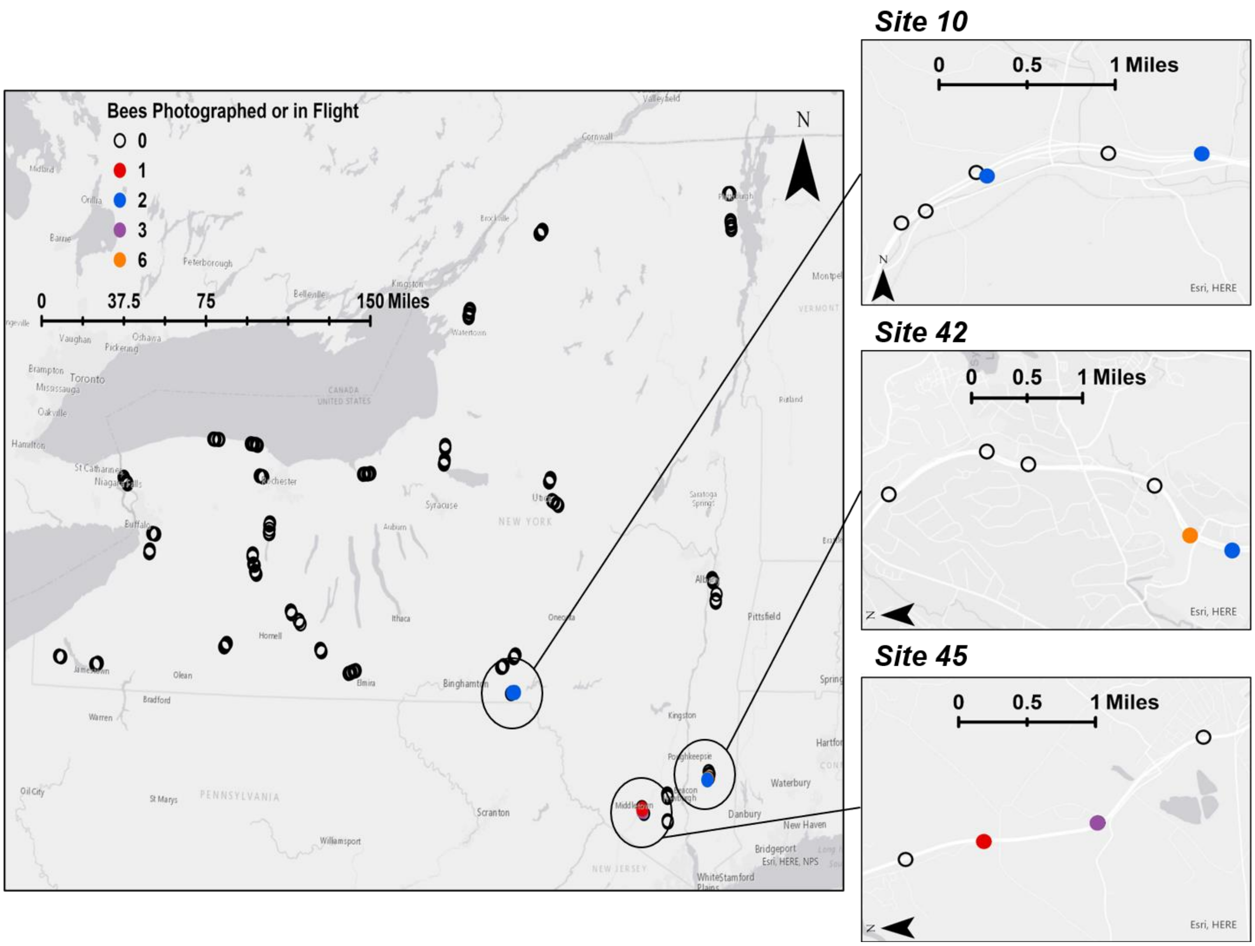


Figure 5: A map with the number of individual bumble bees observed during each photo observation (n=332) Of these observations, six observation periods yield one or more individual bumble bees to be photographed or recorded as in flight. 10 bumble bees were photographed, and six bumble bees were observed in flight. The three highway stretches that bumble bees were observed at are shown in the three small maps (Sites 10, 42, and 45). to the right of the map of all sampling locations. Inset maps of these sites more clearly show the presence of multiple sampling locations with observations and the different number of bees observed at them.

Table 2: Results from photographing bumble bees in 2020. The number of bees photographed (n=10) as well as bees that were observed but not photographed (n=6) and the associated plant family for each are listed.

Plant Family Bumble Bee was Observed on	**Number of Bumble Bees Photographed**	**Number of Bumble Bees in Flight**
Asteraceae	2	4
Fabaceae	0	1
Caprifoliaceae	2	0
Caryophyllaceae	2	0
Hypericaceae	1 *Photograph taken, bee not identified	0
Lamiaceae	2	0
Lythraceae	1	1

Discussion

I was unable to test whether the interaction between roadside mowing and road traffic influence bumble bee abundance. This was due to the lack of bumble bees observed through photography and sweep netting. In the case of the bumble bees caught in the sweep net, a few samples were collected during both field seasons. However, only one bumble bee was caught during the entirety of the 2020 field season. Photography data from 2020 was similarly limited.

It's possible that the addition of other collection methods could have increased the number of non-zero observations. Pan trapping and sweep netting are common methods used to sample bee populations. Due to the nature of this study, pan trapping was not a suitable method. Pan traps are best used when left out for several hours at a given site (Grundel et al., 2011; Popic et al., 2013; Westphal et al., 2008). Sites were spread out across the state so only a limited amount of time was allotted to each sampling location in order to visit each site's replicates multiple times per field season. I visited each sampling location twice per field season, so it was possible that a month or more could have passed in between visits. Using pan traps would have limited the ability to travel across the state, instead favoring local sites. It was critical to examine ROWs across New York because of the large presence of high traffic roads, a feature that is not present in many studies that explore mowing and roadsides.

Instead of pan trapping, I adapted a method for photographing the bees from Loffland et al. (2017) and Cole et al. (2020). Sampling of bee populations produces the best results when combined with another method, so because pan trapping was not possible, a different method was explored (Grundel et al., 2011; Westphal et al., 2008). Richardson et al. (2019) used submitted photographs of bumble bees taken along roadsides in Vermont for identification; however, a specific method was not outlined for how to conduct photography samples. Cole et al. (2020) and Loffland et al. (2017) caught and photographed bumble bees in montane environments as a survey method. Their methods were adapted to work in terms of a much smaller roadside environment. The decreased size of the sampling location may have made the method not as suitable in ROWs, contributing to the lack of bees observed. Development of a method for surveying bees in roadsides that can be completed while at the site without requiring a follow up visit could allow bumble bees to remain a focal species while expanding distance traveled to reach study sites. More studies are needed on traffic and roadside management along large spatial scales as well as with expanded focal organisms such as bumble bees. However, it is possible that bumble bees are not using ROWs and are not present to be detected.

Of the few bumble bees I did see and catch via sweep netting and photography, all were common eastern bumble bees (*B. impatiens*). *B. impatiens* has been observed to be increasing in its range across North America (Cameron et al., 2011; Colla & Packer, 2008; Richardson et al., 2019). *B. impatiens* have been managed for usage in crop pollination since the 1990s (Colla, 2016; Richardson et al., 2019). Directed human support on such a large scale likely contributed to some of the population growth. Managed bees such as *A. mellifera and B. impatiens* are more prone to carrying pathogens, so if/when managed bees escape, wild populations become at risk and could decline in population (McNeil et al., 2020). Furthermore, *B. impatiens* is able to tolerate and live within developed environments better than other bumble bees (Colla & Packer, 2008). Thus potentially this is why they were the only species I observed, as the disturbed nature of roadside ROWs makes them a species that would be able to tolerate these conditions. More sensitive bumble bee species may not have been able to tolerate the disturbed roadside environment and thus made them absent from my observations.

In the case of the bumble bees photographed, two of the three stretches of highway that they were found at were high traffic roads. Bumble bees were observed at site 45, the stretch of road that had the highest traffic of the 30 highway stretches (AADT=53043). Similarly, site 42 was also at the upper bound of traffic with an AADT of 42526 vehicles per day. The other site that bumble bees photographed, as well as the sites that sweep samples we collected from were either not high traffic roads, or contained a mix of traffic levels, ranging from 172 to 25639 vehicles per day. The usage of high traffic roads as well as roads with variable traffic further indicate the ability of *B. impatiens* to tolerate disturbances.

However, the presence of one species of bumble bee does not mean that roadside ROWs are habitats where bumble bees can thrive. There are over 250 species of bumble bees in the world, 18 of which are found in New York (Cornell University, n.d.), so only finding one species of *Bombus* represents a small proportion of actual bumble bee diversity (Kozmus et al., 2011; Williams et al., 2008). The absence of other less common *Bombus* species, particularly ones that are more sensitive to disturbances indicate that roadsides may not be ideal habitat. Low species richness has also been observed in roadsides in Maryland where only 20% of 430 bee species that are known to live in the state. It is possible that the vast collection of disturbances hinders the diversity and abundance of varied and uncommon species in roadsides My research focused on comparing different properties of roadside environments, but it is possible that differences in these characteristics don't contribute to the presence/absence of bumble bees as much as other disturbances.

The small number of bumble bees observed in this study as well as the limited amount of current literature on bumble bees in roadside environments indicate the need for further research into this topic. Specifications as to minimize or manage disturbances in roadside environments could prove useful to bumble bees and other insect pollinators. Additionally, having only observed one species of bumble bee does not provide enough information as to how different *Bombus* species would respond to reduced mowing and traffic in ROW habitats. Understanding if there is an interaction between disturbances such as road traffic and roadside mowing could lead to the identification of areas of high conservation value. Prioritization of conservation areas for critical native insect pollinators that are in decline, like the many species of bumble bees, will be essential in order to maintain ecosystem services and biodiversity in the face of unending human development and ongoing climate change.

Bibliography

Ahrné, K., Bengtsson, J., & Elmqvist, T. (2009). Bumble Bees (Bombus spp) along a Gradient of Increasing Urbanization. PloS One, 4(5). https://doi.org/oi:10.1371/ journal.pone.0005574

Buri, P., Humbert, J. Y., & Arlettaz, R. (2014). Promoting pollinating insects in intensive agricultural matrices: Field-scale experimental manipulation of hay-meadow mowing regimes and its effects on bees. PLoS ONE, 9(1), e85635. https://doi.org/10.1371/journal.pone.0085635

Bartomeus, I., Ascher, J. S., Wagner, D., Danforth, B. N., Colla, S., Kornbluth, S., & Winfree, R. (2011). Climate-associated phenological advances in bee pollinators and bee-pollinated plants. Proceedings of the National Academy of Sciences of the United States of America, 108(51), 20645–20649. https://doi.org/10.1073/pnas.1115559108

Buri, P., Humbert, J. Y., & Arlettaz, R. (2014). Promoting pollinating insects in intensive agricultural matrices: Field-scale experimental manipulation of hay-meadow mowing regimes and its effects on bees. PLoS ONE, 9(1), e85635. https://doi.org/10.1371/journal.pone.0085635

Cameron, S. A., Lozier, J. D., Strange, J. P., Koch, J. B., Cordes, N., Solter, L. F., & Griswold, T. L. (2011). Patterns of widespread decline in North American bumble bees. Proceedings of the National Academy of Sciences of the United States of America, 108(2), 662–667. https://doi.org/10.1073/pnas.1014743108

Cole, J. S., Siegel, R. B., Loffland, H. L., Elsey, E. A., Tingley, M. B., & Johnson, M. (2020). Plant Selection by Bumble Bees (Hymenoptera: Apidae) in Montane Riparian Habitat of California. Environmental Entomology, 49(1), 220–229. https://doi.org/10.1093/ee/nvz159

Colla, S. R. (2016). Status, Threats and Conservation Recommendations for Wild Bumble Bees (Bombus spp.) in Ontario, Canada: A Review for Policymakers and Practitioners. Natural Areas Journal, 36(4), 412–426. https://doi.org/10.3375/043.036.0408

Colla, S. R., & Packer, L. (2008). Evidence for decline in eastern North American bumblebees (Hymenoptera: Apidae), with special focus on Bombus affinis Cresson. Biodiversity and Conservation, 17(6), 1379–1391. https://doi.org/10.1007/s10531-008-9340-5

Colla, S., Richardson, L., & Williams, P. (2011). Guide to bumble bees of the eastern United States. The Xerces Society for Invertebrate Conservation, 104.

Cornell University (n.d.). Species List - Bees of New York [Dataset]. Retrieved from https://pollinator.cals.cornell.edu/sites/pollinator.cals.cornell.edu/files/shared/documents/Bees-of-NY.xlsx

Davis, A. K., Schroeder, H., Yeager, I., & Pearce, J. (2018). Effects of simulated highway noise on heart rates of larval monarch butterflies, danaus plexippus: Implications for roadside habitat suitability. Biology Letters, 14(5). https://doi.org/10.1098/rsbl.2018.0018

Esri Inc., (2021). ArcGIS Pro (Version 2.7.2). Esri Inc. https://www.esri.com/en-us/arcgis/products/arcgis-pro/.

Gardiner, M. M., Riley, C. B., Bommarco, R., & Öckinger, E. (2018, April 1). Rights-of-way: a potential conservation resource. Frontiers in Ecology and the Environment, Vol. 16, pp. 149–158. https://doi.org/10.1002/fee.1778

Grundel, R., Frohnapple, K. J., Jean, R. P., & Pavlovic, N. B. (2011). Effectiveness of Bowl Trapping and Netting for Inventory of a Bee Community. Environmental Entomology, 40(2), 374–380. https://doi.org/10.1603/en09278

Halbritter, D. A., Daniels, J. C., Whitaker, D. C., & Huang, L. (2015). Reducing Mowing Frequency Increases Floral Resource and Butterfly (Lepidoptera: Hesperioidea and Papilionoidea) Abundance in Managed Roadside Margins. Florida Entomologist, 98(4), 1081–1092. https://doi.org/10.1653/024.098.0412

Hopwood, J. L. (2008). The contribution of roadside grassland restorations to native bee conservation. Biological Conservation, 141(10), 2632–2640. https://doi.org/10.1016/j.biocon.2008.07.026

Hopwood, J., Black, S., & Fleury, S. (2015). Roadside Best Management Practices that Benefit Pollinators. 94. Retrieved from https://www.environment.fhwa.dot.gov/env_topics/ecosystems/Pollinators_Roadsides/BMPs_pollinators_landscapes_hi-rez.pdf

Keilsohn, W., Narango, D. L., & Tallamy, D. W. (2018). Roadside habitat impacts insect traffic mortality. Journal of Insect Conservation, 22(2), 183–188. https://doi.org/10.1007/s10841-018-0051-2

Khalid, N., Hussain, M., Young, H. S., Boyce, B., Aqeel, M., & Noman, A. (2018). Effects of road proximity on heavy metal concentrations in soils and common roadside plants in Southern California. Environmental Science and Pollution Research, 25(35), 35257–35265. https://doi.org/10.1007/s11356-018-3218-1

Kozmus, P., Virant-Doberlet, M., Meglič, V., & Dovč, P. (2011). Identification of Bombus species based on wing venation structure. Apidologie, 42(4), 472–480. https://doi.org/10.1007/s13592-011-0037-5

Loffland, H. L., Polasik, J. S., Tingley, M. W., Elsey, E. A., Loffland, C., Lebuhn, G., & Siegel, R. B. (2017). Bumble bee use of post-fire chaparral in the central Sierra Nevada. Journal of Wildlife Management, 81(6), 1084–1097. https://doi.org/10.1002/jwmg.21280

MacPhail, V. J., Richardson, L. L., & Colla, S. R. (2019). Incorporating citizen science, museum specimens, and field work into the assessment of extinction risk of the American Bumble bee (Bombus pensylvanicus De Geer 1773) in Canada. Journal of Insect Conservation, 23(3), 597–611. https://doi.org/10.1007/s10841-019-00152-y

McCleery, R. A., Holdorf, A. R., Hubbard, L. L., & Peer, B. D. (2015). Maximizing the wildlife conservation value of road right-of-ways in an agriculturally dominated landscape. PLoS ONE, 10(3), 1–19. https://doi.org/10.1371/journal.pone.0120375

McFrederick, Q. S., & LeBuhn, G. (2006). Are urban parks refuges for bumble bees Bombus spp. (Hymenoptera: Apidae)? Biological Conservation, 129(3), 372–382. https://doi.org/10.1016/j.biocon.2005.11.004

McNeil, D. J., McCormick, E., Heimann, A. C., Kammerer, M., Douglas, M. R., Goslee, S. C., … Hines, H. M. (2020). Bumble bees in landscapes with abundant floral resources have lower pathogen loads. Scientific Reports, 10(1), 1–12. https://doi.org/10.1038/s41598-020-78119-2

Munguira, M. L., & Thomas, J. A. (1992). Use of Road Verges by Butterfly and Burnet Populations, and the Effect of Roads on Adult Dispersal and Mortality. The Journal of Applied Ecology, 29(2), 316. https://doi.org/10.2307/2404501

Muñoz, P. T., Torres, F. P., & Megías, A. G. (2015). Effects of roads on insects: a review. *Biodiversity and Conservation*, *24*(3), 659–682. https://doi.org/10.1007/s10531-014-0831-2

New York State Department of Transportation. (2015). Annual Average Daily Traffic (AADT) Overview [PDF]. Retrieved from https://data.ny.gov/Transportation/Annual-Average-Daily-Traffic-AADT-Beginning-1977/6amx-2pbv

New York State Department of Transportation. (2017). Highway Design Manual [PDF]. Retrieved from https://www.dot.ny.gov/divisions/engineering/design/dqab/hdm/hdm-repository/chapt_14.pdf

New York State Department of Transportation. (2019). Traffic Data Viewer Geodatabase Shapefile [Dataset]. Retrieved from https://www.dot.ny.gov/divisions/engineering/applications/traffic-data-viewer/traffic-data-viewer-repository/NYSDOT_TDV_AADT_2019.gdb.zip

Nichols, R. N., Goulson, D., & Holland, J. M. (2019). The best wildflowers for wild bees. Journal of Insect Conservation, 23(5–6), 819–830. https://doi.org/10.1007/s10841-019-00180-8

Noordijk, J., Delille, K., Schaffers, A. P., & Sýkora, K. V. (2009). Optimizing grassland management for flower-visiting insects in roadside verges. Biological Conservation, 142(10), 2097–2103. https://doi.org/10.1016/j.biocon.2009.04.009

Ogilvie, J. E., Griffin, S. R., Gezon, Z. J., Inouye, B. D., Underwood, N., Inouye, D. W., & Irwin, R. E. (2017). Interannual bumble bee abundance is driven by indirect climate effects on floral resource phenology. Ecology Letters, 20(12), 1507–1515. https://doi.org/10.1111/ele.12854

Phillips, B. B., Gaston, K. J., Bullock, J. M., & Osborne, J. L. (2019). Road verges support pollinators in agricultural landscapes, but are diminished by heavy traffic and summer cutting. *Journal of Applied Ecology*, *56*(10), 2316–2327. https://doi.org/10.1111/1365-2664.13470

Phillips, B. B., Wallace, C., Roberts, B. R., Whitehouse, A. T., Gaston, K. J., Bullock, J. M., … Osborne, J. L. (2020). Enhancing road verges to aid pollinator conservation: A review. Biological Conservation, 250(July), 108687. https://doi.org/10.1016/j.biocon.2020.108687

Popic, T. J., Davila, Y. C., & Wardle, G. M. (2013). Evaluation of Common Methods for Sampling Invertebrate Pollinator Assemblages: Net Sampling Out-Perform Pan Traps. PLoS ONE, 8(6). https://doi.org/10.1371/journal.pone.0066665

Pyke, G. H., Inouye, D. W., & Thomson, J. D. (2011). Activity and abundance of bumble bees near Crested Butte, Colorado: Diel, seasonal, and elevation effects. Ecological Entomology, 36(4), 511–521. https://doi.org/10.1111/j.1365-2311.2011.01295.x

R Core Team. (2020). R: A Language and Environment for Statistical Computing. Vienna, Austria. http://www.r-project.org

Richardson, L. L., McFarland, K. P., Zahendra, S., & Hardy, S. (2019). Bumble bee (Bombus) distribution and diversity in Vermont, USA: a century of change. Journal of Insect Conservation, 23(1), 45–62. https://doi.org/10.1007/s10841-018-0113-5

Ries, L., Debinski, D. M., & Wieland, M. L. (2001). Conservation value of roadside prairie restoration to butterfly communities. Conservation Biology, 15(2), 401–411. https://doi.org/10.1046/j.1523-1739.2001.015002401.x

Russell, K. N., Russell, G. J., Kaplan, K. L., Mian, S., & Kornbluth, S. (2018). Increasing the conservation value of powerline corridors for wild bees through vegetation management: an experimental approach. Biodiversity and Conservation, 27(10), 2541–2565. https://doi.org/10.1007/s10531-018-1552-8

Saarinen, K., Valtonen, A., Jantunen, J., & Saarnio, S. (2005). Butterflies and diurnal moths along road verges: Does road type affect diversity and abundance? *Biological Conservation*, *123*(3), 403–412. https://doi.org/10.1016/J.BIOCON.2004.12.012

Seitz, N., vanEngelsdorp, D., & Leonhardt, S. D. (2019). Conserving bees in destroyed landscapes: The potentials of reclaimed sand mines. Global Ecology and Conservation, 19, e00642. https://doi.org/10.1016/j.gecco.2019.e00642

Skórka, P., Lenda, M., Moroń, D., Kalarus, K., & Tryjanowski, P. (2013). Factors affecting road mortality and the suitability of road verges for butterflies. Biological Conservation, 159, 148–157. https://doi.org/10.1016/j.biocon.2012.12.028

Soluk, D. A., Zercher, D. S., & Worthington, A. M. (2011). Influence of roadways on patterns of mortality and flight behavior of adult dragonflies near wetland areas. Biological Conservation, 144(5), 1638–1643. https://doi.org/10.1016/j.biocon.2011.02.015

Suzuki-Ohno, Y., Yokoyama, J., Nakashizuka, T., & Kawata, M. (2017). Utilization of photographs taken by citizens for estimating bumblebee distributions. Scientific Reports, 7(1), 1–11. https://doi.org/10.1038/s41598-017-10581-x

Nichols, R. N., Goulson, D., & Holland, J. M. (2019). The best wildflowers for wild bees. Journal of Insect Conservation, 23(5–6), 819–830. https://doi.org/10.1007/s10841-019-00180-8

Thomson, D. M. (2016). Local bumble bee decline linked to recovery of honey bees, drought effects on floral resources. Ecology Letters, 19(10), 1247–1255. https://doi.org/10.1111/ele.12659

Vanbergen, A. J., Garratt, M. P., Vanbergen, A. J., Baude, M., Biesmeijer, J. C., Britton, N. F., …

Wright, G. A. (2013). Threats to an ecosystem service: Pressures on pollinators. Frontiers in Ecology and the Environment, 11(5), 251–259. https://doi.org/10.1890/120126

Villemey, A., Jeusset, A., Vargac, M., Bertheau, Y., Coulon, A., Touroult, J., … Sordello, R. (2018). Can linear transportation infrastructure verges constitute a habitat and/or a corridor for insects in temperate landscapes? A systematic review. Environmental Evidence, 7(1), 1–33. https://doi.org/10.1186/s13750-018-0117-3

Volenec, Z. M., & Dobson, A. P. (2019). Conservation value of small reserves. Conservation Biology, 0(0), 1–14. https://doi.org/10.1111/cobi.13308

Ward, K., D. Cariveau, E. May, M. Roswell, M. Vaughan, N. Williams, R. Winfree, R. Isaacs, and K. Gill. 2014. Streamlined Bee Monitoring Protocol for Assessing Pollinator Habitat. 16 pp. Portland, OR: The Xerces Society for Invertebrate Conservation.

Westphal, C., Bommarco, R., Carré, G., Lamborn, E., Morison, N., Petanidou, T., … Steffan, I. (2008). Measuring Bee Diversity in Different European Habitats and. Ecological Monographs, 78(4), 653–671.

Williams, P. H., Cameron, S. A., Hines, H. M., Cederberg, B., & Rasmont, P. (2008). A simplified subgeneric classification of the bumblebees (genus Bombus). Apidologie, 39(1), 46–74. https://doi.org/10.1051/apido:2007052

Williams, N. M., Cariveau, D., Winfree, R., & Kremen, C. (2011). Bees in disturbed habitats use, but do not prefer, alien plants. Basic and Applied Ecology, 12(4), 332–341. https://doi.org/10.1016/j.baae.2010.11.008

Williams, P. H., & Osborne, J. L. (2009). Bumblebee vulnerability and conservation world-wide. Apidologie, 40(3), 367–387. https://doi.org/10.1051/apido/2009025

Winfree, R., Aguilar, R., Vázquez, D. P., LeBuhn, G., & Aizen, M. A. (2009). A Meta-Analysis of Bees' Responses to Anthropogenic Disturbance. Ecology, 90(8), 2068–2076. from http://www.jstor.org/stable/25592723

Wojcik, V., & Buchmann, S. (2012). Pollinator Conservation And Management On Electrical Transmission And Roadside Rights - Of - Way : A Review. Journal of Polination Ecology, 7(3), 16–26.

Wood, T. J., Michez, D., Paxton, R. J., Drossart, M., Neumann, P., Gérard, M., … Vereecken, N. J. (2020). Managed honey bees as a radar for wild bee decline? Apidologie, 51(6), 1100–1116. https://doi.org/10.1007/s13592-020-00788-9

Wrzesien, M., & Denisow, B. (2016). Distribution and abundance of bee forage flora across an agricultural landscape - Railway embankments vs. road verges. Acta Societatis Botanicorum Poloniae, 85(3), 1–15. https://doi.org/10.5586/asbp.3509

Supplemental Data

Supplemental Data File 1: **Bumble bee photography data**

https://figshare.com/s/357db56a30ef6ab101de