8-2019

The Effects of Urbanization on Avian Seed Dispersal Success of Toxicodendron radicans (Anacardiaceae)

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Recommended Citation

Stanley, Amber M., "The Effects of Urbanization on Avian Seed Dispersal Success of Toxicodendron radicans (Anacardiaceae)" (2019). *Electronic Theses and Dissertations.* Paper 3619. https://dc.etsu.edu/etd/3619

The Effects of Urbanization on Avian Seed Dispersal Success of *Toxicodendron radicans* (Anacardiaceae)

A thesis

presented to

the faculty of the Department of Biological Sciences

East Tennessee State University

In partial fulfillment

of the requirements for the degree

Master of Science in Biology

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by

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August 2019

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Keywords: Urbanization, *Toxicodendron radicans*, seed dispersal, plant-animal interactions, ornithochory

ABSTRACT

The Effects of Urbanization on Avian Seed Dispersal Success of *Toxicodendron radicans* (Anacardiaceae)

by

Amber Stanley

Urbanization is increasing at a dramatic rate as the human population increases. While it is well-known that urbanization tends to decrease species diversity (i.e., biotic homogenization), it is not known how urbanization affects the frequency and efficiency of species interactions. Seed dispersal is a plant-animal interaction that depends on disperser feeding rate, disperser diversity, probability of seed dispersal and germination. How these factors are affected by urbanization however is unknown. In this study, we evaluate how urbanization alters these factors. Urban sites had 2x higher feeding rate and 3x higher number of disperser species. The probability of seed dispersal however was the same between natural and urban sites. Moreover, the probability of germination after dispersal was 20% lower in urban sites, leading to overall negative effects of urbanization on *T. radicans* seed dispersal. In this study we demonstrated that urbanization can affect species diversity, as well as their ecological functions.

ACKNOWLEDGMENTS

I would like to thank Dr. Arceo-Gomez, Dr. Alsop, Dr. Pyles, and Dr. Karsai for agreeing to be on my committee. Thank you all for your encouragement, patience, and guidance.

I would also like to give credit my labmates who assisted me in this project. Thanks are due to Daniel Barker, Jesse Daniels, and Cheril Patel for risking poison ivy rashes to help me, when I know you were itching to be anywhere else. Stephen Duncan, Rachel Herrmann, and Amelia Schroeder (in addition to the three mentioned above), I thank for helping me assemble seed traps. Thank you also to all of my friends in the Department of Biology who supported me.

Thanks are also due to Kevin Brooks for assisting me in data collection during the second field season. At least we didn’t get shot at.

Thank you to the Department of Biological Sciences for the Graduate Teaching Assistantship; without which I could not have joined the Master’s program. I am also grateful for the teaching experience, and I have learned many invaluable lessons from it.

Thank you to the Department of Biological Sciences at ETSU for the Marcia Davis Research Grant. Thank you to the School of Graduate Studies at ETSU for the School of Graduate Studies Research Grant. Thank you to the Knoxville Chapter of the Tennessee Ornithological Society for the Marcia Davis Memorial Grant. Without these funds, I would not have been able to conduct a second field season.

Thank you to Chris Stoehrel of the Forest Service for helping me gain permission to use sites in the Cherokee National Forest. Thank you to Riki Forney and the Town of Erwin for giving me permission to use the Erwin site. Thank you to Dr. Darrell Moore for permission to use the old Marine Corps Armory as an urban site.

Thank you to my family, who supported me from the sidelines during this journey. A special thanks to my three roommates (they know who they are) who put up with my busy schedule and provided the means of transportation to all of my study sites. I love you guys!

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CHAPTER 1

INTRODUCTION

Urbanization

During the 20th century, the growth of the human population reached almost exponential levels, expanding from 1.6 billion people (global population estimate for the year 1900) to over 6 billion in the year 2000 (UN DESA 2017). Estimates project the human population will reach approximately 10 billion people by 2050 (UN DESA 2017). As the human population grows, people tend to concentrate in urban areas. For instance, as the global population expanded in the 20th century, so did the rate of immigration into urban areas and growth of urban infrastructure (i.e., urbanization) to accommodate the increasing demand. In fact, the global percentage of urbanization grew from 15% in 1900 to 50% by 2008; this ongoing process is expected to reach 67% by 2050 (Satterthwaite et al. 2010).

The most dramatic examples of urbanization can be found in major metropolitan areas, such as Beijing, China; Cairo, Egypt; Mexico City, Mexico; and New York City, New York, where natural habitat has been almost entirely destroyed (Demographia 2018). Smaller cities are also experiencing urbanization (Zhou et al. 2014); Raleigh, North Carolina, for example, has physically expanded from approximately 83 thousand hectares to approximately 160 thousand hectares in only the past two decades (5.1% expansion annually; "Atlas of Urban Expansion" 2016).

As global urbanization increases, more natural habitat is destroyed, fragmented, or encroached upon, leading to negative effects for the continued survival of natural ecosystems and species that rely on them (Marzluff 1997; McKinney 2006; Trentanovi et al. 2013; Zhou et al. 2014).

Effects of Urbanization

Perhaps the most detrimental effect of urbanization is the destruction and loss of natural habitat, due to increased demand for space and natural resources (Heinrichs et al. 2016). Habitat loss and fragmentation has been implicated as the leading cause of biodiversity loss and species extinction (Pimm and Raven 2000; Butchart et al. 2010; Kiers et al. 2010; Baiser et al. 2012). Total habitat loss, such as clear-cutting or strip-mining, is not as common as habitat fragmentation, where swathes of habitat are separated by roads, agriculture, or other urban infrastructure (Butchart et al. 2010).

Habitat degradation (e.g. less abundant and lower quality resources, lower quality nesting sites and feeding grounds) affects the ability of species to survive (Heinrichs et al. 2016). Degradation is often a side effect of urbanization, due to increased human activity in and around natural areas (Olden et al. 2006). Mechanisms of habitat degradation include environmental stressors such as pollution (antibiotics, noise, light, pesticides), and introduction of nonnative species (Gleditsch and Carlo 2011; Zhang et al. 2011; Gavrilescu et al. 2015; Valiente-Banuet et al. 2015; Knop et al. 2017). Habitat fragmentation and degradation mechanisms do not exist in isolation, and it is often a combination of many of these factors that lead to decreasing trends in species diversity (as reviewed in McKinney 2008; Heinrichs et al. 2016).

Biotic Homogenization

The most studied consequence of urbanization is the alteration of species diversity, both in the number of species (species richness) and how evenly those species are represented in a community (in terms of the number of individuals per species, i.e. species evenness; McKinney and Lockwood 1999; Olden and Rooney 2006; McKinney 2008; Dar and Reshi 2014). While urbanization typically decreases survival of native species, there are often species more able to cope with urban stressors and who tend to be highly competitive generalists. These species are often nonnative and invasive (though not always), and tend to dominate urban habitats, in both numbers and tenacity (McKinney and Lockwood 1999; Olden and Rooney 2006).

The phenomenon, known as biotic homogenization, has been observed throughout a wide range of taxa. For example, in the United States and Canada, one study estimated the overall amount of biotic homogenization for plants, freshwater fishes, reptiles, amphibians, mammals, and birds (Olden et al. 2006), finding that each group experienced a decrease in species diversity as a result of urbanization. The degree to which these taxa have been affected, however, varies by location within North America (Olden et al. 2006). White et al. (2018) compared the structure and diversity of avian communities in Britain from 1960-70 to 1980-90 and found that species diversity was homogenized with fewer species, but a greater number of individuals of the species present.

Most studies focus on changes in taxonomic diversity (number and representation of species), however, in the past twenty years there has been a shift to examining functional diversity by investigating the number and representation of species interactions (Tobias and Monika 2012; White et al. 2018). For example, Tobias and Monika (2012) found functional diversity tended to homogenize along with taxonomic diversity in a natural forest system, although not at the same rate. Similar effects have been found in soil microbial communities in the tropics, and among pollinator groups such as bees, birds, butterflies, and moths (Winfree et al. 2011; Pauw and Louw 2012; Rodrigues et al. 2013; Deguines et al. 2016). While species diversity is most commonly studied, changes in functional diversity cannot be inferred from changes in taxonomic diversity alone, and thus species interactions should be studied in addition to species diversity (Tobias and Monika 2012).

Species Interactions

Species do not exist in isolation. Instead, they engage in both intraspecific and interspecific interactions. Intraspecific interactions include competition for resources, reproduction, child-rearing, and others, amongst members of the same species (Des Roches et al. 2018). Interspecific interactions include competition, predation, consumption of resources, symbiotic relationships, and others, among members of different species (Adler et al. 2018).

The focus of this study is a specific plant-animal interaction. These interactions can be mutualistic (in which both members benefit), commensal (in which one member benefits, but the other is not affected positively or negatively), parasitic (in which one member benefits while the other loses), or competitive (in which both members lose) (Adler et al. 2018). Plant-animal interactions are an integral component of well-functioning ecosystems and contribute to such key functions as nutrient recycling, pollination, and seed dispersal (McConkey and O'Farrill 2016).

Plant-Animal Interactions—Seed Dispersal

Seed dispersal by animal vectors is a mutualistic or commensal interaction by which a plant's offspring are moved away via ingestion by or attachment to an animal vector (Schupp 1993). Seed dispersal allows a mother plant to increase its reproductive fitness and reduce competition by dispersing offspring into favorable habitat at a distance from the mother plant (Levey et al. 2008). This process also increases the genetic diversity of habitats by allowing immigration of genetic material from more distant plants (McConkey and O'Farrill 2016).

Without seed dispersal, plants may experience a lower reproductive fitness, lack of gene flow, inbreeding depression, and increased competition for resources (McConkey and O'Farrill 2016; Jordano 2017). Successful seed dispersal depends on many factors, including the rate of interactions with dispersers, i.e., number of feeding events over time; the diversity of disperser species, which can alter the success of the seed post-dispersal; the probability of seed dispersal from an interaction; and the probability of germination post-dispersal (Schupp 1993).

While studies addressing the effects of urbanization often focus on species diversity, there are fewer studies that examine the effects of urbanization on the diversity, intensity and efficiency of species interactions (i.e., functional diversity) within a community (Tylianakis et al. 2008; Valiente-Banuet et al. 2015; McConkey and O'Farrill 2016). This study attempts to address this gap by evaluating the effects of urbanization on the seed dispersal interactions between *Toxicodendron radicans* and its community of avian seed dispersers.

Questions & Hypotheses

This study addresses the following question: "What are the effects of urbanization on the dynamics of avian seed dispersal for *Toxicodendron radicans*?"

In order to investigate this question, four specific areas were addressed:

1) Will the frequency of bird feeding events on *T. radicans* fruits be different in urban sites compared to natural sites?

Hypothesis 1: More feeding will occur in urban sites due to the openness of the habitat and the visibility of food resources.

Hypothesis 2: Alternatively, less feeding will occur in urban sites due to the anthropogenic disturbance decreasing the abundance and activity of birds.

Hypothesis 3: There will be no difference between the feeding rate of birds in urban and natural sites.

2) How will the composition of the disperser community differ between urban and natural sites?

Hypothesis 1: There will be a greater diversity of avian species in natural sites, due to the effects of biotic homogenization in urban sites.

Hypothesis 2: Alternatively, a greater number of avian species will be found interacting with *T. radicans* in urban sites, owing to the open habitat and more easily visible food sources in urban systems.

3) Will the probability of seed dispersal be different between urban and natural sites?

Hypothesis 1: Seeds will be dispersed with a greater probability in urban sites due to a more visible resource availability attracting a greater number of feeding birds.

Hypothesis 2: Seeds will be dispersed with a lower probability in urban sites due to a lower feeding rate and lower abundance of dispersers as a result of anthropogenic disturbances.

4) Will germination of avian-defecated *T. radicans* differ between urban and natural sites?

Hypothesis 1: Due to the effects of biotic homogenization, the expected disperser community in urban sites will be comprised of generalist species. These species will be unable to adequately prepare the seeds for germination due to differences in their gastrointestinal tract physiology, leading to a lower probability of germination of defecated seeds from birds in urban sites.

Hypothesis 2: Alternatively, the differences in gut physiology of different disperser species may have no effect on germination, so long as the seed experiences acid scarification within the gut.

CHAPTER 2

MANUSCRIPT FORMATTED FOR SUBMISSION TO ECOLOGY

Urbanization negatively affects avian seed dispersal success of *Toxicodendron radicans* (Anacardiaceae)

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ABSTRACT

Urbanization is increasing at a dramatic rate as the human population increases. While it is well-known that urbanization tends to decrease species diversity (i.e., biotic homogenization), it is not known how urbanization affects the frequency and efficiency of species interactions. Seed dispersal is a plant-animal interaction that depends on several factors for success, including disperser feeding rate, disperser diversity, and the probability of seed dispersal and germination. In this study, we evaluate how urbanization alters the above factors that affect the success of seed dispersal. Urban sites tended to have two times higher feeding rate and three times higher number of species. The probability of seed dispersal, however, was the same between natural and urban sites. The probability of germination, however, was 20% lower in urban sites, leading to overall negative effects of urbanization on *T. radicans* seed dispersal. In this study, we demonstrated that urbanization can not only affect species diversity, but also affect their ecological functions, in this case seed dispersal.

Key words: urbanization, ecological functions, plant-animal interactions, seed dispersal, ornithochory, *Toxicodendron radicans*.

INTRODUCTION

The rate of global urbanization is increasing at an alarming pace as the human population has grown from 1.6 to 6.1 billion people in the past century (Satterthwaite et al. 2010; UN DESA 2017). The resulting increase in the human population has required the expansion of urban habitat, (e.g. buildings, roads, and other urban infrastructure; Satterthwaite et al. 2010; Zhou et al. 2014) as the percentage of people living in urban areas has increased from 15% to 50%. This accelerated rate of urbanization is expected to increase as we reach over 9 billion people by 2050, 67% of which are expected to reside in urban areas (Satterthwaite et al. 2010; UN DESA 2017). As urbanization expands, the surrounding natural habitat is modified through increased habitat loss, degradation, and fragmentation, and species invasion—all having detrimental effects on native species diversity (McKinney and Lockwood 1999; Pimm and Raven 2000; Butchart et al. 2010; Kiers et al. 2010). As a result, there is an urgent need to understand the effects of growing urbanization on the persistence and stability of both native species and natural ecosystems as a whole.

The loss of species diversity has been documented as one of the major effects of increasing urbanization on the environment (McKinney and Lockwood 1999; Sax and Gaines 2003; Kiers et al. 2010). In urban areas, one or a few species tend to dominate while other species are lost, and this reduction in overall diversity is known as biotic homogenization (McKinney 2006; Baiser et al. 2012). For example, White et al. (2018) found that the diversity of avian species homogenized as urbanization increased in Great Britain over a twenty-year period. Other studies have shown similar results for plants (Trentanovi et al. 2013), fish (Leitao et al.

2016), pollinators (Winfree et al. 2011), ants (Holway and Suarez 2006), and soil-dwelling bacteria (Rodrigues et al. 2013). Furthermore, the few species dominating urban habitats are often generalists, and while they interact with many other species, these interactions are typically weak and/or inefficient (McKinney and Lockwood 1999; Tobias and Monika 2012). While the effects of urbanization on species diversity have been well-documented, our understanding of how urbanization affects the frequency, intensity and efficiency of species interactions is still poorly understood. Altering the efficiency or frequency of species interactions can lead to the loss of important ecological functions, with negative consequences for individual species and the stability of the ecosystem (McDonnell et al. 1997; Valiente-Banuet et al. 2015).

Animal-based seed dispersal can be particularly susceptible to the effects of urbanization (Ruxton and Schaefer 2012; McConkey and O’Farrill 2016). Successful seed dispersal depends on several factors, including the frequency of interactions with dispersers and the diversity of the disperser community, which affects both the probability of a seed being dispersed and the probability of a seed germinating after dispersal (Schupp 1993). The frequency of interactions with dispersers (i.e., the number of feeding events per time interval) can alter the success of seed dispersal, with a higher frequency of interactions increasing the potential for seeds to be dispersed (Schupp 1993). The diversity of disperser species can also affect successful seed dispersal. Different disperser species have different feeding behaviors and foraging distances, thus affecting seed fate and dispersal distances (Levey et al. 2008; Carlo et al. 2013) and mediating the dispersal of seeds into habitats favorable for germination (Leitao et al. 2016; Sebastián-González 2017; Martin-Albarracin et al. 2018). Additionally, disperser species can have a direct impact on the germinability of seeds, because some seeds must travel through the gastrointestinal tract of the disperser (Traveset 1998; Traveset et al. 2001) requiring acid and/or mechanical scarification in order to break this physical dormancy (Baskin and Baskin 2014). The amount of acid and mechanical scarification is often dependent upon the gut physiology of the disperser species (Murphy et al. 1993; Traveset et al. 2001). Animal dispersers who have co-evolved with specific plants tend to have the optimal conditions for scarification of seeds within their guts (Karasov and Levey 1990). Additionally, animal dispersers that primarily consume fruits and seeds tend to have an optimal gut transit time (from ingestion to excretion), which is much shorter than the transit time of other foods, such as insects (Karasov and Levey 1990).

Animal-based seed dispersal in urban environments can be impeded by several factors, including decreased disperser diversity and efficiency due to habitat fragmentation and disturbance. For instance, optimal foraging theory predicts that dispersers should favor food resources that are less costly to find and obtain. (Brown 1988). In this sense, studies have found that urban habitats are able to support a greater number of birds due to a greater availability of food resources and a lower predation rate (McKinney and Lockwood 1999; Bolger 2001; Marzluff and Ewing 2001), which can lead to a higher frequency of feeding in urban habitats (Marzluff 1997). A higher frequency of feeding may in turn lead to a higher seed dispersal probability. Furthermore, urbanization typically provides conditions that favor generalist and invasive species, while more specialized species tend to be extirpated (McKinney and Lockwood 1999; Olden and Rooney 2006). Plants may thus experience a decrease in effective seed dispersal due to suboptimal scarification by generalist and nonnative disperser species (Levey and Karasov 1994). These disturbances combined can have important implications for the continuing persistence of plant species (McConkey and O’Farrill 2016). For example, interruption of seed dispersal has been shown to lead to the loss of overall connectivity between plant populations (Neuschulz et al. 2016). Loss of population connectivity means that there is no flow of individuals or genetic material between or among populations, which can lead to decreased survival, reduction of genetic diversity, inbreeding depression, and increased risk of extinction (McConkey and O'Farrill 2016; Jordano 2017). Seed dispersal is also an important component of habitat connectivity, which helps to buffer against the negative consequences of disturbances, such as urbanization (Neuschulz et al. 2016). However, the effects of urbanization on the frequency, diversity, or efficiency of seed dispersal have been little studied, and thus, it is not yet well-known whether, or how, urbanization alters the dynamics of plant-seed disperser interactions.

This study compares the dynamics of seed dispersal between natural and urban sites using *Toxicodendron radicans* and its avian seed dispersing community as a model system. I specifically ask the following questions: 1) Is frequency of bird feeding events on *T. radicans* fruits higher in urban sites compared to natural sites? 2) Is the composition of the seed disperser community different between urban and natural sites? 3) Is the probability of seed dispersal different between urban and natural sites? 4) Is germination of avian-defecated *T. radicans* seeds collected from urban sites lower than those collected from natural sites?

METHODS

Study Organism

The impacts of urbanization on avian seed dispersal were studied using Eastern Poison Ivy (*Toxicodendron radicans*) as the focal species. *Toxicodendron radicans* is a common plant in the Eastern United States (Gillis 1971) that can have several growth forms (ground cover, shrub, liana). This study focused only on lianas (woody vines that affix themselves to trees) because they produce the most amount of fruits (A. Stanley, pers. obs). The plant is dioecious (separate male and female individuals; Gillis 1971), and the female can produce up to thousands of fruits each year (A. Stanley, pers. obs.). The fruits are drupes (single-seeded), drab in coloration (pale green when unripe, off-white when ripe), waxy, and represent a high energy food source (47% lipid content; Warawdekar and Jannke 1957; Gillis 1971; Cazetta et al. 2008). Fruits ripen from August-September and can persist into March (Robinson and Handel 1993). The primary consumers of the seeds are birds of the orders Passeriformes (perching birds) and Piciformes (woodpeckers) (Martin et al. 1951). Squirrels have also been recorded eating *T. radicans* seeds, however, they destroy the seeds during consumption (Krefting and Roe 1949). While some studies have documented which species consume poison ivy seeds (Martin et al. 1951), no comprehensive study of *T. radicans* seed dispersers has been published. The seeds require a minimum cold stratification time of ~3 weeks for germination (Schiff et al. 2004). More importantly, the seeds require mechanical and acid scarification—via digestion in the avian gut—, in order to successfully germinate (Schiff et al. 2004). Thus, changes in the diversity and composition of the seed disperser community may influence recruitment in this species. Poison ivy is a common plant in both natural and urban ecosystems, where it typically grows in disturbed and open habitats, as well as secondary forests (Gillis 1971), and is therefore an ideal system to evaluate the effects of urbanization on seed dispersal dynamics.

Study Sites

Over the course of two field seasons (September 2017- March 2018, October 2018- January 2019), five urban sites and four natural sites where poison ivy was present were studied (Fig. 2.1). Natural sites were located in the Cherokee National Forest, in Washington and Carter counties of Tennessee (Fig. 2.1, top row). Urban sites were located in Johnson City, Tennessee (Washington County) and Erwin, Tennessee (Unicoi County) (Fig. 2.1, bottom row).

Figure 2.1: *Field Sites.* Satellite images with latitude longitude coordinates of all sites used over two years. Sites with a bold blue border were used in the first field season. Natural sites (top row) have a high percentage of tree cover and few roads intersecting the habitat. Urban sites (bottom row) have little tree cover and many roads fragmenting the habitat.

The average distance between sites was 14.72 km (range 5.26-33.53 km) (Fig. 2.2). Urban and natural sites were determined by estimating the percentage of urban infrastructure (buildings, roads, railroads, sidewalks, etc.) in a 1-km radius using i-Tree Canopy (v6.1) (Fig. 2.3). Sites with more than 30% infrastructure were considered urban (McKinney 2002). Natural sites were characterized by a high percentage of tree cover and less than 5% infrastructure (McKinney 2002). Urban sites used in this study still contained an average 23% tree cover and over 30% of other natural surfaces (primarily grass); without this cover, these urban sites would not be able to support the *T. radicans* plants that birds exploit. One natural site was used in both the first and second field seasons, which served as a qualitative gauge of the differences between the two seasons. The remaining sites were only visited in the course of a single field season. Sites were visited throughout the season until there was no longer any poison ivy fruits to observe.

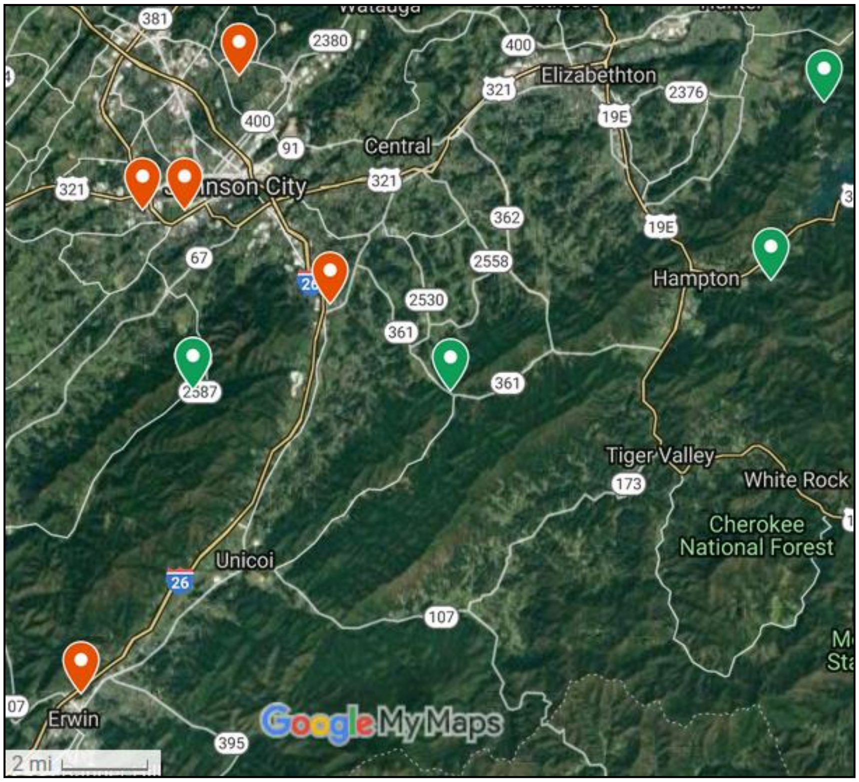

Figure 2.2: *Map of Field Sites.* Google map indicating the location of all field sites used over two years. Natural sites (green markers) were located within the Cherokee National Forest, in Washington and Carter counties. Urban sites (orange markers) were located within Washington and Unicoi counties.

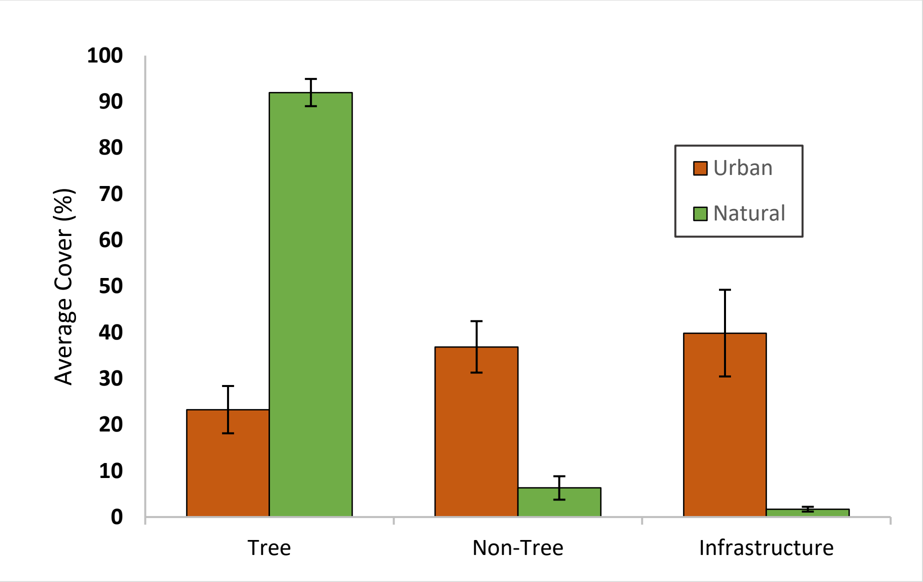

Figure 2.3: *Average Percent Cover of Natural and Urban Sites*. Average percent cover of sites and standard error, calculated with i-Tree Canopy. The Non-Tree category represents all natural surfaces other than trees such as rocks, water, grass, and shrubs. The Infrastructure category represents all urban surfaces such as roads, railroads, buildings, and sidewalks.

Feeding Rate and Disperser Community Composition

To estimate the rate of seed disperser feeding on poison ivy plants in urban versus natural sites, 5-7 individual lianas were selected at each site. Lianas selected within a site were attached to different trees, in an attempt to observe genetically distinct individuals instead of asexually reproduced clones. Each liana was observed for 1-3 periods of 12 minutes during each visit to a site. Visits to a site were conducted during the active feeding time of birds at each site, as determined by three all-day visits to each site over the course of the field season. During each observation period, the identity and quantity of feeding avian species was recorded, as well as the time of day that the interaction occurred. From these data, feeding rate was estimated as the number of birds feeding per 12-minute observation period. An average feeding rate was estimated for each individual liana for each visit.

To account for differences in resource availability (number of fruits) among sites, the total number of fruits available was estimated at each site at the beginning, middle, and end of the fruiting season. This was performed by summing the total number of fruits attached to each liana used in observations.

Probability of Seed Dispersal

To estimate the probability of seed dispersal at urban and natural sites 91-1422 (1-10%) of fruits from 5-7 individual lianas were marked at eight sites (2 sites in the first field season, 6 sites in the second field season), with a UV fluorescent dye (Llewellyn Data Processing LLC) following methods from Levey and Sargent (2000) (Fig. 2.4). Although some birds are able to perceive part of the UV spectrum (Levey and Sargent 2000), one study found that there was no effect of UV-marking on feeding preference (Willson and Whelan 1989). To confirm this result, preliminary food preference was conducted to test for this effect. One hundred UV-marked and 100 unmarked seeds were mixed into a tray and left outside for birds to feed on for 4 hours, during the morning when birds are actively feeding. The leftover seeds were collected and examined for the UV-marking. This test was repeated 10 times, totaling 2000 seeds tested.

Unmarked and marked seeds under normal light

Unmarked and marked seeds under blacklight

Figure 2.4: *Unmarked and UV Fluorescent Marked Fruits*. A fruit marked with UV fluorescent dye (top) and an unmarked fruit (bottom) under ambient light (left) and under blacklight (right). Note that the UV dye cannot be discerned by the human eye without the aid of a blacklight.

Seed traps (comprised of mesh netting over a PVC frame) were placed under marked fruits on the liana, to collect the fallen fruits that were not dispersed (Fig. 2.5). This methodology has been shown to be highly efficient at catching and retaining non-dispersed fallen fruits (Stevenson and Vargas 2008). This methodology was co-opted for this study, using a light mesh material with a pore size >5mm.

Figure 2.5: *Seed Trap*. Placed underneath the marked fruits of an individual liana. Traps were made from a mesh netting sewn over PVC frames. This construction allowed the falling fruits to be retained by the trap, instead of bouncing off.

Fruits caught in the traps were collected up to three times a week and checked for presence of the UV dye, ripeness, and viability in the lab. For each liana, the number of dispersed seeds was estimated as:

Seeds dispersed = Total fruits marked – Marked fruits found in trap

From this equation, the probability of seed dispersal was calculated by dividing the total number of marked fruits by the number of seeds dispersed.

Differences in Germination Between Natural and Urban Sites

To evaluate differences in germination of dispersed poison ivy seeds from natural and urban sites, we conducted a germination experiment using seeds collected from avian feces and undigested seeds collected at natural and urban sites (from seed traps). Specifically, seeds from four treatments were germinated: 1) undigested seeds soaked in water for 1 hour before planting (negative control), 2) undigested seeds soaked in sulfuric acid—an artificial scarification method reported to induce the highest germination frequency in *T. radicans* (Benhase and Jelesko 2013)—for thirty minutes before planting (positive control; Tilki and Bayraktar 2013), 3) dispersed seeds collected from Natural, and 4) Urban sites. All seeds were cold stratified at 4^{o}C for 30-90 days. Five seeds of a single treatment were planted in a seed square, in propagation mix soil (Sungro Horticulture). Between 39-43 squares of each treatment were then placed in random order within a tray and placed inside a growth chamber for five weeks. Conditions for the first week were 24 hours darkness at ambient temperature. The light dark cycle for the remaining four weeks was 12:12 L:D, and the temperature a constant 25^{o}C (Schiff et al. 2004). Squares were watered as needed to keep the soil moist. Germination was scored once per day over the course of five weeks. A germination event was recorded once the radicle had emerged from the seed, and the date of germination was recorded.

Once the five-week trial was completed, all non-germinated seeds were checked for viability by dissection (Fig. 2.6), as other members of the family Anacardiaceae have been shown to produce a high proportion of nonviable seeds (González-Varo et al. 2018). From this, a proportion of viable seeds germinated was produced for each square using the equation:

Proportion germinated = Viable seeds germinated / Total # viable seeds in square

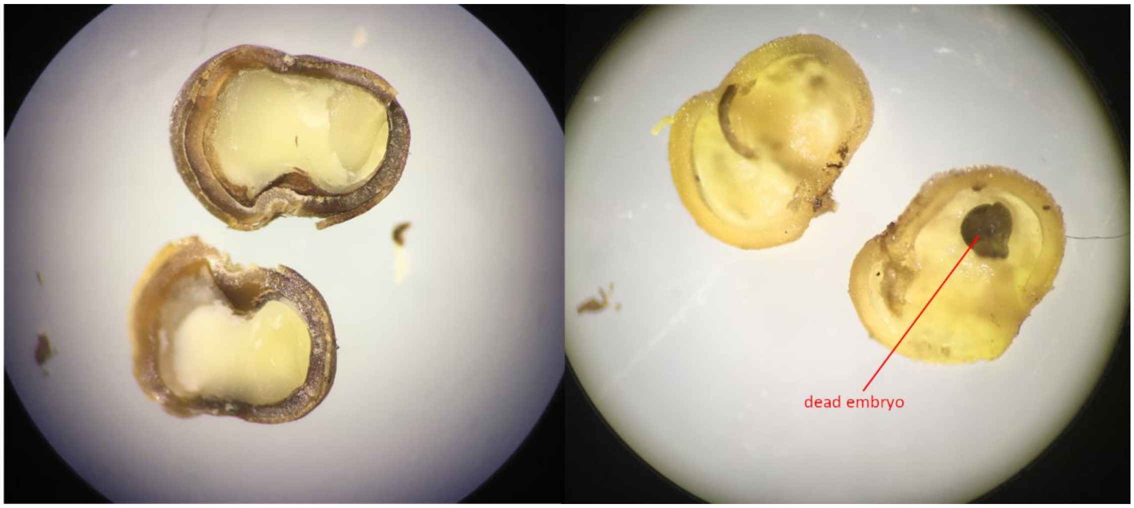

Figure 2.6: *Viable and Nonviable Dissected Seeds*. Dissection of viable (left) and non-viable (right) *T. radicans* seeds. Notice that nonviable seeds no longer contain large amounts of endocarp. Instead, a dead embryo lies inside the hollowed seed.

A total of 815 seeds were tested for germinability: 210 defecated seeds collected from natural sites, 210 defecated seeds from urban sites, 200 non-defecated seeds treated with water, and 195 non-defecated seeds treated with sulfuric acid.

To test for differences in baseline seed germination between urban and natural sites, an experiment was conducted, in which non-defecated seeds from natural sites (n=139) and urban sites (n=140) were scarified in sulfuric acid and grown for 5 weeks under the conditions stated above.

Data analysis

Species accumulation curves were produced using EstimateS (v9.1.0) based on observation data to estimate sampling effort during both field seasons (Colwell et al. 2004).

To test for any effect of the UV fluorescent dye on the feeding preference of birds, a t-test was used to compare the number of leftover UV-marked and unmarked seeds from 10 trials.

In order to test for differences in feeding rate between natural and urban sites, a generalized mixed model (GLMM) was used. Site category (natural or urban) was considered a fixed effect. Random effects were site (nested within category), plant, visit number (hereafter called round) and year. Resource availability in a particular site at a given time period was included as a covariate. A lognormal distribution was used which better fit the residuals of the model.

For differences in species richness of the seed disperser community GLMM with a Poisson distribution was used. Site category (urban vs natural) was a fixed effect, and site (nested within category), plant, and year were random effects. To compare the overall similarity of disperser species composition between natural and urban sites, we used Morisita's Index of Overlap (Wolda 1981). It takes into account the number and abundance of species and uses the formula:

$$\mathbf{C_D = (2\, \Sigma x_i y_i)/(D_x + D_y)XY,}$$

Where x_i is the number of times species i is found in site X from one visit, y_i is the number of times species i is found in site Y from another visit, and Dx and Dy are the Simpson's diversity indices for samples x and y.

The Morisita Index ranges from 0 (no similarity between sites) to 1 (sites are completely similar), and can be expressed as a percentage (Wolda 1981).

To compare the probability of seed dispersal between natural and urban sites, we first transformed the response variable, using a logit transformation, to meet assumptions of normality of the residuals. We then used a mixed model with site category (urban vs natural) as a fixed effect, site (nested within category), trap, and year as random effects.

To compare the germination rate in sulfuric acid-treated seeds, water-treated seeds, and defecated seeds from natural and urban sites, we used a mixed model with treatment type as a fixed effect and tray, square (nested within tray), and trial as random effects. We used a Tukey-adjusted least square means post-hoc test to compare the treatments to each other.

RESULTS

Sampling Effort

Rarefaction curves show that sampling effort was sufficient to observe 85.6% of species in urban sites and 79.2% of species in natural sites (Fig. 2.7). In the first field season, both urban sites were visited 14 times each (31.2 hours and 44 hours observation time), one natural site was visited 18 times (39.8 hours), and the other natural sites was visited 10 times (20.8 hours). In the second field season, all urban sites were visited 6 times (8 hours observation time each), one natural site was visited 4 times (5 hours), and two natural sites were visited 5 times each (9 hours each).

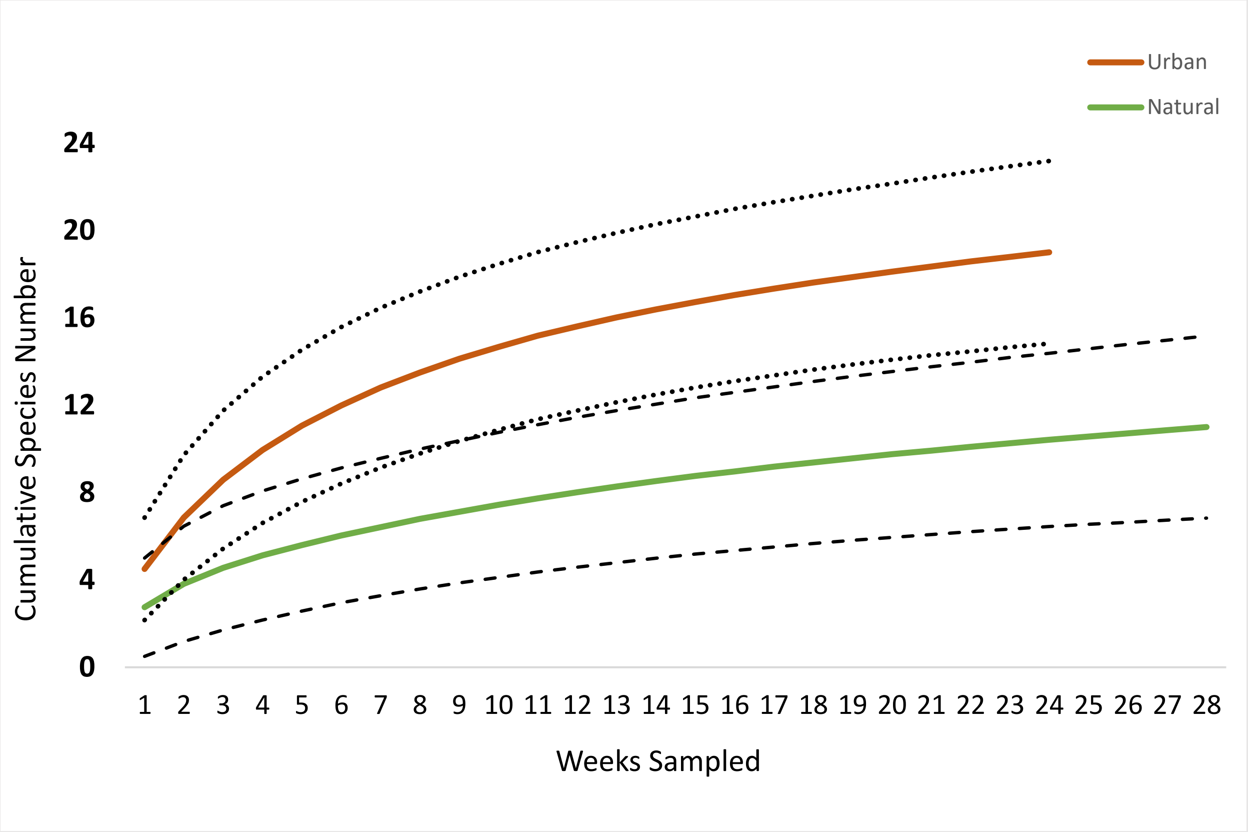

Figure 2.7: *Species Accumulation Curves.* Species accumulation curves for each site used in the first field season. The orange line represents the curve for urban sites, with the dotted black lines representing its 95% confidence interval. The green line represents the curve for natural sites, with the dashed black lines representing its 95% confidence interval.

Resource Availability

In the first field season, urban sites contained an average of 19,144 fruits and natural sites contained an average of 9,506 fruits. In the second field season, urban sites contained an average of 6,066 fruits and natural sites contained an average of 2,170 fruits.

Seed Marking

A preliminary food preference test showed that presence or absence of the UV dye did not affect the food choice (*P* = 0.847; Fig. 2.8). Thirty seed traps were deployed during the study, 15 in urban sites and 15 in natural sites. The traps remained until there were no UV marked fruits on the liana. 9,500 fruits were marked, 5,762 in urban sites, 3,738 in natural sites. Over 21,000 seeds were collected in total; 14,220 from urban sites, and 7762 from natural sites; 6,166 fruits were collected in the first season, and 15,690 were collected in the second field season. Of the collected seeds, 1,617 were marked and 2,866 were defecated.

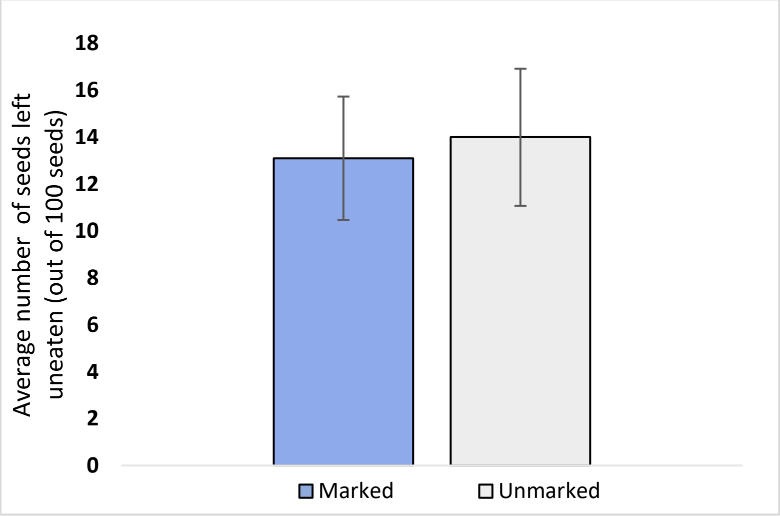

Figure 2.8: *Average Number of Leftover Seeds from Feeding Trials*. Results from ten trials of a preference test using 100 UV-marked and 100 unmarked sunflower seeds randomly mixed in a feeding tray per trial. At the end of each trial, the total number of marked and unmarked seeds left uneaten were counted. Average and standard error are shown.

Feeding Rate

We recorded a total of 415 feeding events, 333 feeding events in urban areas (99.2 hours of observation), and 82 feeding events in natural sites (83.6 hours). Feeding rate (number of feeding events per observation interval) was significantly different between natural and urban sites (*F* = 11.61, *P* < 0.01). Feeding rate was 2.3x higher in urban compared to natural sites (Fig. 2.9).

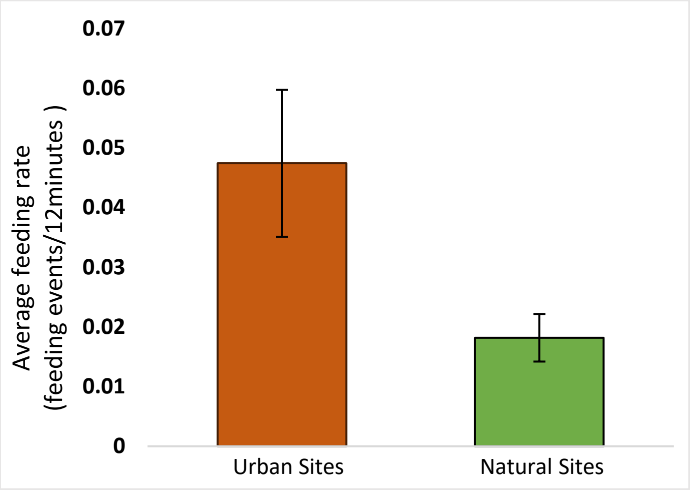

Figure 2.9: *Average Feeding Rate in Natural and Urban Sites*. Average number of feeding events (with standard error bars) per 12-minute interval of avian species on *T. radicans* in natural and urban sites. Feeding rate is 2.3x higher in urban sites.

Disperser Community Composition

A total of twenty-three avian species were observed feeding on poison ivy (see Appendix A). Twenty species were found in urban sites, while eleven species were found in natural sites. Of these species, twelve were found only in urban sites, and three were found only in natural sites. Average seed disperser species richness was significantly higher in urban sites (13.8 ± 4.6) than in natural sites (4.3 ± 1.3) (*F* = 5.46, *P* = 0.048). Yellow-rumped Warblers (*Dendroica* *coronata*) and Northern Cardinals (*Cardinalis cardinalis*) were the most frequent feeders in urban sites, representing 55.9% of observed feeding events (Fig. 2.10a). Carolina Chickadees (*Poecile carolinensis*) and Downy Woodpeckers (*Picoides pubescens*) were the most frequent feeders in natural sites, representing 72.0% of feeding events (Fig. 2.10b). The calculated Morisita Index of Overlap revealed a 77.8% similarity in community composition between urban and natural sites.

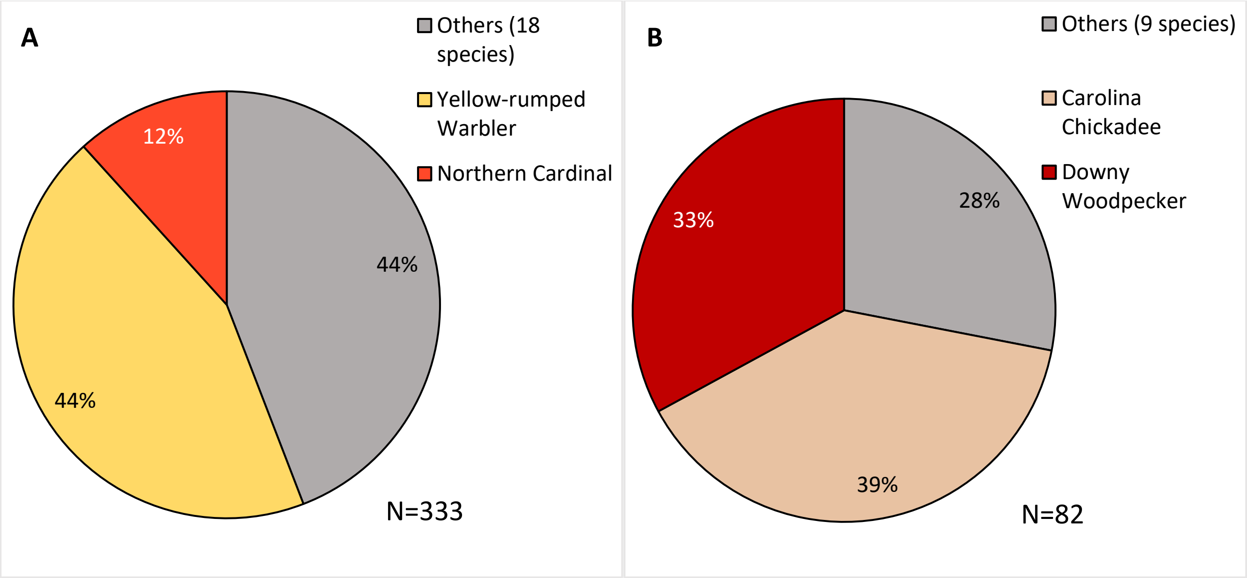

Figure 2.10: *Feeding Event Distribution by Species in Natural and Urban Sites.* **A)** Distribution of feeding events in urban sites, with focus on the two most abundant species, Yellow-rumped Warblers (147 out of 333 feeding events) and Northern Cardinals (39 out of 333 feeding events). **B)** Distribution of feeding events in natural sites, with focus on the two most abundant species, Carolina Chickadee (32 out of 82 feeding events), and Downy Woodpecker (27 out of 82 feeding events).

Probability of Seed Dispersal

Of the 9,500 fruits marked, only 1,617 fruits were collected from seed traps. We collected 905 fruits from urban sites and 712 from natural sites. Both urban and natural sites had a high probability of seed dispersal (>80%), and the probability was not significantly different between them (Figure 2.11, $F = 0.17$, $P = 0.7$).

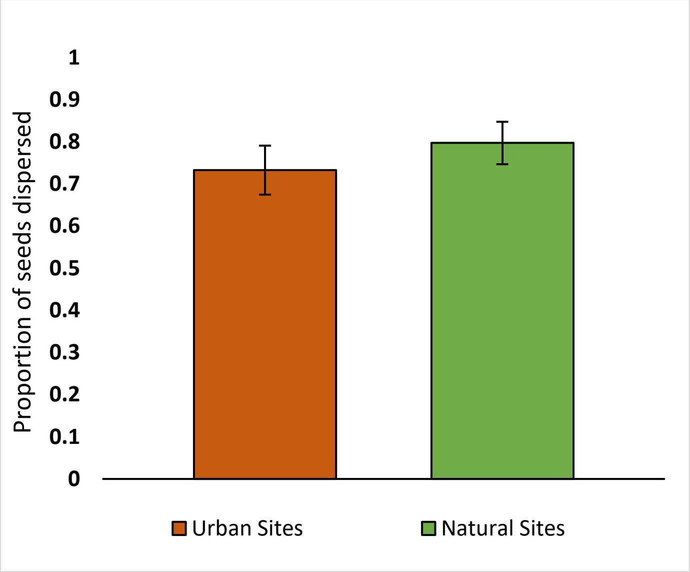

Figure 2.11: *Average Probability of Seed Dispersal in Natural and Urban Sites*. Average and standard error are shown. Probability of seed dispersal was greater than 70% in both natural and urban sites.

Differences in Germination

Baseline germination of non-defecated seeds from natural and urban sites were not significantly different from each other ($P = 0.48$; Fig. 2.12). The rate of germination differed significantly between treatments of the main germination experiment ($F = 20.20$, $P < 0.0001$). The germination rate of defecated seeds collected from natural sites was significantly different from that of defecated seeds collected from urban sites (Table 2.1). Germination is 20% higher in defecated seeds collected from natural sites than in urban seeds (Fig. 2.13). Neither natural nor urban defecated seeds had a germination rate significantly different from sulfuric acid-treated seeds (Table 2.1). All other treatments had a germination rate significantly different from the water-treated seeds (Table 2.1).

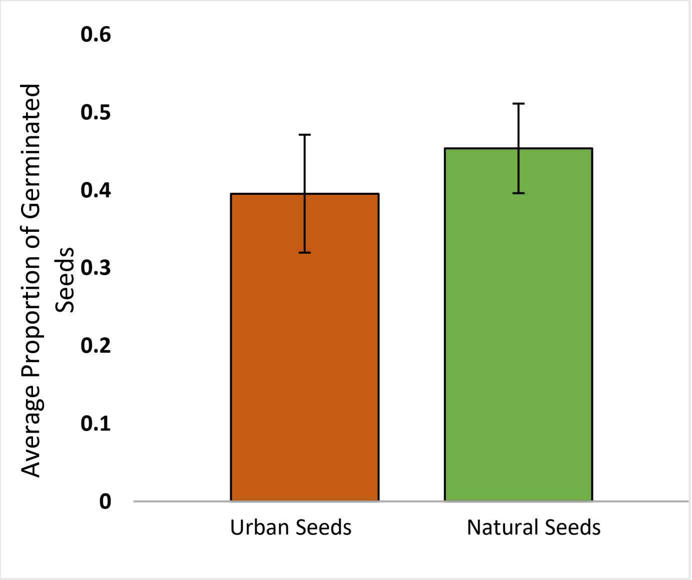

Figure 2.12: *Baseline Germination of Natural and Urban Seeds*. Average proportion (with standard error bars) of germinated non-defecated seeds collected from natural and urban sites, then treated with sulfuric acid.

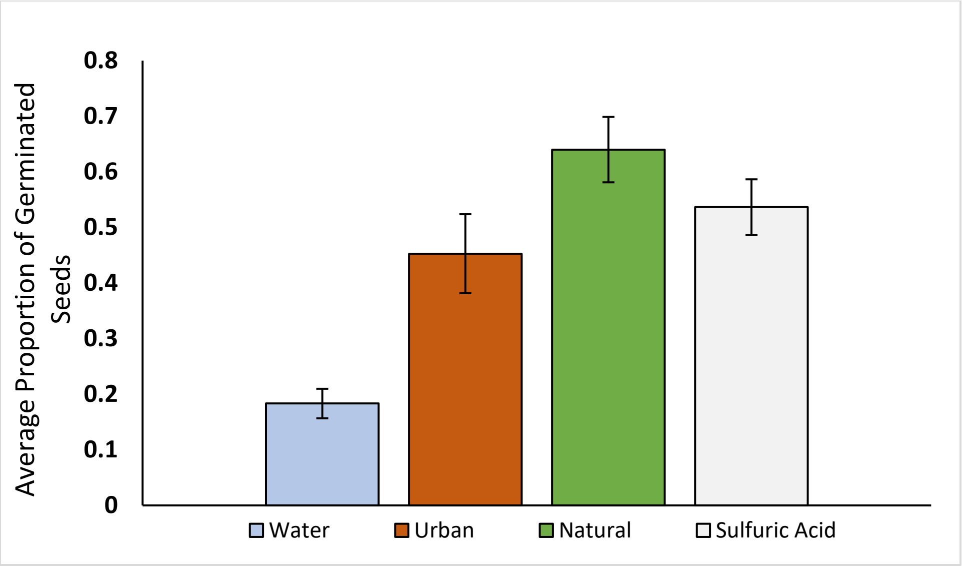

Figure 2.13: *Average Proportion of Germinated Seeds from Four Treatments*. Average proportion and standard error are shown. Defecated seeds collected from natural sites had a germination rate 20% higher than those collected from urban sites. Natural and urban defecated seeds were not significantly different from the sulfuric acid treated seeds. Natural, urban, and sulfuric acid treatments had a germination rate significantly higher than water treated seeds.

Table 2.1: *Post-hoc comparison of water treatment to other treatments*. Tukey-adjusted post-hoc comparison of the treatments in the main germination experiment.

Treatment Comparisons	**P-value**
Natural defecated & Sulfuric Acid	**0.42**
Natural defecated & Urban defecated	**0.03**
Natural defecated & Water	**<0.0001**
Sulfuric Acid & Urban defecated	**0.64**
Sulfuric Acid & Water	**<0.0001**
Urban defecated & Water	**<0.0001**

DISCUSSION

Urbanization is a global phenomenon that converts natural habitats into less habitable urban landscapes. As the human population continues to increase, so does the rate of urbanization. The effects of urbanization are most often studied in terms of species diversity, however, urbanization can also alter ecosystem functions, by impacting the frequency and efficiency of interactions between organisms (Pauw and Louw 2012; Deguines et al. 2016).

Feeding Rate

In this study, we found that urbanization increases the frequency of interactions; we observed a 2x higher feeding rate in urban sites than in natural sites, which is consistent with our first hypothesis. This may be mediated in part by resource availability, as we also found 2x the number of poison ivy fruits in urban sites. Other studies have shown that a greater quantity of resources attracts a higher number of individuals, and an increased feeding rate as a result (Davidar and Morton 1986; Kwit et al. 2004a; Gleditsch and Carlo 2011). One study found that poison ivy is particularly receptive to the increased carbon dioxide (CO_2) concentrations found in urban systems, showing an 149% increase in growth compared to poison ivy exposed to CO_2 levels found in natural systems (Mohan et al. 2006). Urban sites may allow poison ivy plants to produce more fruits to entice dispersers, as studies have shown that other plants in urban settings produce more energy to allocate toward fruit production than plants in natural settings (Ziska et al. 2004; Lambrecht et al. 2016; Zhao et al. 2016). However, we accounted for resource availability in our mixed model analysis and found that it is not the only factor driving differences in feeding rate between urban and natural sites. Urban sites are more open habitats (>30% tree cover), which may allow birds to spot resources easier than in natural sites. Additionally, urban sites may be more limited in the variety of other food resources, as studies have shown that fewer fruiting species are commonly found in urban areas (DeCandido 2004; Belaire et al. 2014). This may increase the consumption of poison ivy by birds, if other, more preferable food resources are scarce.

Disperser Community Composition

Our study found that urban sites tended to have a 3x greater species richness than natural sites, consistent with our second hypothesis for this question. However, urban sites also tended to have a single species that dominated feeding interactions during observation. For example, Yellow-rumped Warblers were observed feeding on *T. radicans* 107 times in a single urban site over one field season; the second-most frequent feeder in that site were Northern Cardinals, observed feeding only 34 times in the same season. While species richness was higher in urban sites, the functional diversity of urban sites was negatively impacted; we found a greater number of generalist and omnivorous species in urban sites than in natural sites, in which we observed primarily over-wintering frugivorous species (see Appendix A). Similar results have been found in metanalyses of pollinator communities, where generalist species tended to take over plant-pollinator interactions and functional diversity tended to become homogenized (Pauw and Louw 2012; Deguines et al. 2016).

Probability of Seed Dispersal

In this study, we found that the probability of a seed being dispersed was not significantly different between urban and natural sites, a result inconsistent with both hypotheses. The differences in disperser species between urban and natural sites (3x higher species richness in urban sites) did not seem to affect the probability of a seed being dispersed. While there was a 2x higher feeding rate in urban sites, there was also a 2x higher resource availability to support this increased feeding rate. This high seed dispersal probability (greater than 70%) is accredited to the dispersal vector, as birds are efficient dispersal agents in either habitat (Levey et al. 2008; Carlo et al. 2013). However, a metanalysis found that anthropogenic disturbance (such as hunting and logging) tended to decrease the number of seeds being dispersed, particularly in tropical systems, a result contrary to ours (Markl et al. 2012). The severity of disturbance in tropical systems may not support seed dispersal processes. While urbanization did not have a significant effect on this aspect, there was a negative effect on the outcome of seed dispersal in our study.

Differences in Germination

Seeds dispersed by birds in urban sites were 20% less likely to germinate than seeds dispersed by birds in natural sites, consistent with our first hypothesis. This effect is not due to any inherent differences in the seeds themselves, as there was no significant difference in the probability of germination of non-defecated seeds from urban and natural sites. Rather, it is likely due to the differences in exposure to acid scarification and the time a seed spends in the gastrointestinal tract of different species. This may be due to the different dietary preferences of each bird, as studies have shown that birds of a primarily frugivorous (or switch to a frugivorous diet in the winter season) diet tend to have a shorter gut transit time of seeds (~20-30 minutes) compared to omnivorous species' gut transit time (~60 minutes), with the optimal gut transit time of seeds ranging from 25-40 minutes (Karasov and Levey 1990, 1992, 1994). Urban sites had a greater number of species (and individuals of those species) that maintain an omnivorous/generalist diet, such as European Starlings (*Sturnus vulgaris*). This change in disperser species, due to urbanization, have altered the efficiency of seed dispersal in urban sites.

This study has shown that urbanization negatively affects avian seed dispersal success of *T. radicans*. If this study had focused solely on measuring species diversity (e.g. most biotic homogenization studies; White et al. 2018) or feeding rate (e.g. most pollination studies; King et al. 2013), our results would have suggested that urbanization has a positive effect on this seed dispersal interaction. However, by examining species diversity, interaction frequency, as well as the efficiency of the interaction, we find a different story. Additionally, we were able to pinpoint the potential mechanism underlying our results: the change in composition of the disperser community caused a 20% decrease in germination success in urban sites.

While poison ivy in urban sites tended to produce a greater number of fruits and attracted more dispersers, the 20% reduction in the probability of germination decreased the reproductive success of these plants. This reduction in reproductive success may impact the genetic diversity of the plant population in urban sites. With continued negative effects on reproductive success, the population may experience genetic isolation and a reduced ability to withstand further disturbance (McConkey and O'Farrill 2016; Jordano 2017). While our study focused on a single plant species, there are thousands of other fruiting species across the globe that are facing the pressures of urbanization (Kissling et al. 2009; Silva et al. 2016).

Studies that examine the effects of urbanization focus primarily upon its effects on species diversity (Holway and Suarez 2006; McKinney 2006; Olden et al. 2006; Alberti et al. 2017). It is widely known that urbanization tends to decrease species diversity (biotic homogenization), yet we know this does not tell the whole story. In the past two decades, there has been a shift in focus from species diversity to functional diversity, which tells a more complete story (White et al. 2018). Our study has contributed to the slowly growing body of knowledge about the effects of urbanization on functional diversity. More research focused on the effects of urbanization on functional diversity, rather than solely species diversity, should be performed.

ACKNOWLEDGMENTS

We thank Chris Stoehrel and the Cherokee National Forest for permission to use sites within the Cherokee National Forest. We thank Riki Forney and the town of Erwin, Tennessee for permission to use a site within the town of Erwin. We also thank Dr. Darrell Moore for permission to use an urban site. We also thank the members of the Arceo-Gómez lab and Kevin Brooks for their support and assistance in data collection and seed trap construction. This research was supported by the ETSU Department of Biological Sciences Marcia Davis Grant, ETSU School of Graduate Studies Research Grant, and the Knoxville Chapter of the Tennessee Ornithological Society Marcia Davis Memorial Grant.

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CHAPTER 3

SUMMARY & FUTURE DIRECTIONS

This study examined the effects of urbanization on a plant-animal interaction between *Toxicodendron radicans* and its avian seed dispersal community. While we found a greater feeding rate and a greater number of bird species in urban sites, the probability of seed dispersal was not significantly different between natural and urban sites, and seed collected from urban sites had a 20% lower probability of germination. This lower germination probability is due to the different bird species found dispersing poison ivy seeds in urban sites, and not due to any differences between non-defecated seeds in urban and natural sites (as shown by a baseline germination experiment, where non-defecated seeds from urban and natural sites were subjected to the same treatment conditions and had no significant difference in the probability of germination). In conclusion, this study has shown that urbanization has a negative effect on seed dispersal of our focal plant species by altering its disperser community. As urbanization continues to increase, it is likely that we will see further negative effects on this process, as well as other processes.

This study was limited natural forests and urban/semi-suburban sites. Future studies should address sites with intermediate disturbance, such as suburban and agriculture sites. Another limit to this study was a potential observer effect, which may have altered the behavior of the disperser species. However, efforts were taken to mitigate this effect, by observing from a long distance (approx. 50 feet) with binoculars, minimal talking or noise during observations, and wearing drab colored clothes (with the exception of one site, were the observer had to wear hunter orange garb to prevent incident).

While this study addressed several factors of successful seed dispersal, there were many that were beyond the scope of this study. These factors should be addressed in future work and are discussed below.

In order for seed dispersal to be considered successful, a dispersed seed must be able to germinate and survive the seedling stage (Schupp et al., 2010). For a seed to germinate and survive, it must first be deposited into favorable habitat. One study has found that seeds of a common European weed are up to 55% less likely to land in habitat favorable for germination in urban areas (Cheptou et al., 2008). Additionally, other studies have shown that urban soils tend to contain heavy metals and other pollutants that can affect the germination and survival of seeds (Trombulak and Frissell 2000). To properly compare the outcome of seed dispersal between natural and urban sites, the likelihood of poison ivy seeds being dispersed to favorable habitat for germination or landing on a surface not conducive to survival—i.e., concrete or other impermeable urban surfaces—should be studied.

While seed dispersal is an important plant-animal interaction, there are other interactions that also rely on animals. While it is known that urbanization can homogenize insect diversity (Winfree et al., 2011), we know less about how urbanization affects pollination. One study has shown that pollen transfer of an herbaceous plant in urban areas is as successful as pollen transfer for this species in agriculturally managed habitat (Van Rossum, 2010). However, this study does not compare the success of pollen transfer between urban and natural habitats (Van Rossum, 2010). Poison ivy produce clusters of small whitish-green flowers that bloom for only a few weeks in May-June (A. Stanley pers. obs.). This plant relies on insects to transfer pollen from one individual to another, so it makes sense to evaluate the impact of urbanization on the pollinator community and pollination success in this plant. Pollination of poison ivy in natural and urban sites could be studied by observing pollinator visitation; collecting pollinators to examine their pollen load; and collecting the female plants parts (style and stigma) post-pollinator visit, to microscopically view pollen that has been deposited and sent its genetic information to the ovule of the flower via pollen tubes.

In our study, the second field season had much less feeding on poison ivy, because it was still warm enough to support the persistence of insects, a preferred food source. As climate change continues to cause warmer and shorter winters, poison ivy, and perhaps other winter fruiting species may experience a severe drop in seed dispersal. In addition to warmer temperatures, climate change also causes a shift in migration times and patterns (Mayor et al., 2017). This may also affect feeding patterns, which in turn decreases seed dispersal. One interesting trend was noted in the course of the germination experiments: defecated seeds with longer cold stratification times tended to germinate with a higher probability than seeds that spent less time in cold stratification. When this effect was experimentally tested—natural and urban defecated seeds treated with 3 weeks or 3 months cold stratification time—, we found that this trend was still present, but non-significant.

In the course of the germination experiments, it was noted that when some seeds began to germinate, they would extend a radicle covered in visible roots hairs. These roots hairs, as time passed, would grow thicker and denser and turn a reddish color (Figure 3.1a). Other seeds did not produce these thick red sections of dense root hairs (Figure 3.1b). In one germination trial, it was noted that 50% of the defecated seeds from natural sites grew the denser red patch of root hairs, and only 4.5% defecated seeds from urban sites grew thick root hairs. Studies have shown modulation in root hair density based on availability of nutrients—less nutrients cause the growth of more root hairs to increase nutrient and water uptake—however, all seeds in our germination trials were grown in the same constant conditions (Bahmani et al., 2016; He et al., 2005; Ma et al., 2001; Salazar-Henao et al., 2016). Perhaps these dense root hair patches confer a greater advantage to poison ivy seedlings to grow and survive? This can be tested by germinating non-defecated seeds treated with sulfuric acid, and then identifying individuals with dense root hairs (hairy) and those without dense root hairs (bald). Groups of hairy and bald seedlings could then be transferred into different growing conditions, such as standard soil with regular watering (control), nutrient-deprived soil, water-deprived soil, and both nutrient- and water-deprived soil. Growth could be scored over time by measuring change in stem length, number and size of leaves, and at the end of the growth period, total mass of the seedling.

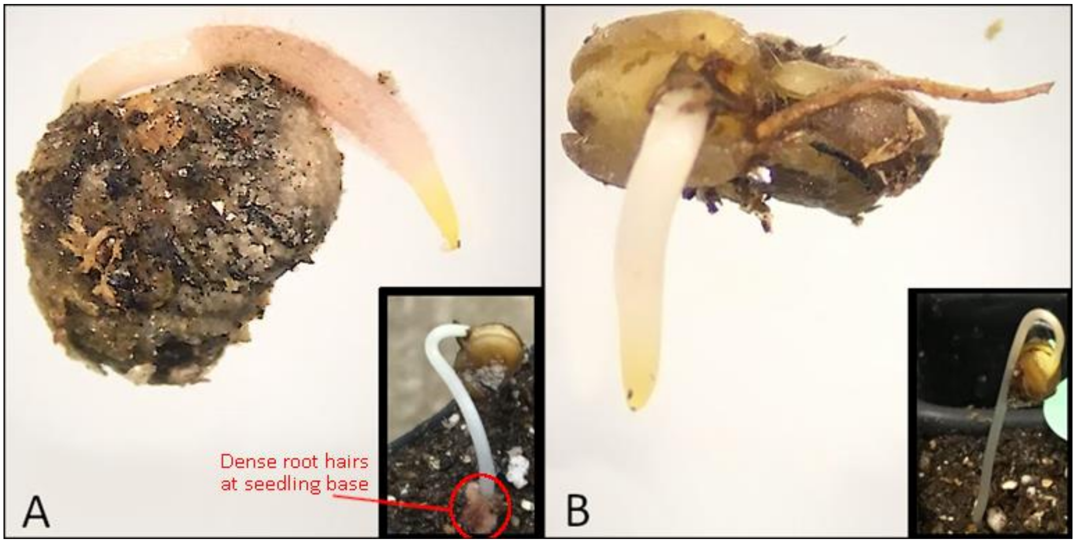

Figure 3.1: *Germinated Seeds With and Without Dense Root Hairs.* Germinated seeds of a similar growth stage with (A) and without (B) dense root hairs, as seen under dissecting microscope at 20x magnification.

While there are many more directions which this research could take, I will end with one final suggestion: Poison ivy is a proliferous plant, able to spread asexually and persist in unfavorable habitats, but what of other, more sensitive plants that rely on sexual reproduction—i.e., seed dispersal—to spread? In East Tennessee, there are many fruiting plants that produce fruits in late fall and winter, such as Holly (*Ilex spp.*) and Flowering Dogwood (*Cornus florida*) trees. Additionally, there are several species of nonnative and invasive fruiting bushes, such as Burning Bush (*Euonymus alatus*), Multiflora Rose (*Rosa multiflora*), and several Honeysuckle species (*Lonicera spp.*) that can be found in East Tennessee. Without limiting this study to East Tennessee, there are thousands of fruiting plants globally that are facing the pressures of urbanization (Kissling et al., 2009; Silva et al., 2016). How does urbanization affect their success? It remains to be studied.

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APPENDICES

Appendix A: List of Birds Observed Feeding on Poison Ivy in this Study

Order	**Family**	**Common Name (*Species name*)**	**Status (Native, Nonnative, Overwintering, Migratory)**	**Primary Diet (Frugivore, Insectivore, Granivore, Omnivore)**	**Frequency of Feeding in Natural Sites**	**Frequency of Feeding in Urban Sites**
Passeriformes	Cardinalidae	Northern Cardinal (*Cardinalis cardinalis*)	Native, Overwintering	Granivore/Frugivore	**2**	**39**
		Rose-breasted Grosbeak (*Pheucticus ludovicianus*)	Native, Migratory	Frugivore/Insectivore	**2**	**0**
	Corvidae	Blue Jay (*Cyanocitta cristata*)	Native, Overwintering	Omnivore	**0**	**4**
	Fringillidae	House Finch (*Haemorhous mexicanus*)	Nonnative, Overwintering	Frugivore/Insectivore	**0**	**4**
	Mimidae	Brown Thrasher (*Toxostoma rufum*)	Native, Migratory	Omnivore	**0**	**1**
		Gray Catbird (*Dumetella carolinensis*)	Native, Migratory	Frugivore/Insectivore	**0**	**2**
		Northern Mockingbird (*Mimus polyglottos*)	Native, Overwintering	Omnivore	**0**	**1**
	Paridae	Carolina Chickadee (*Poecile carolinensis*)	Native, Overwintering	Insectivore/ Frugivore	**32**	**32**
		Tufted Titmouse (*Baeolophus bicolor*)	Native, Overwintering	Insectivore/Frugivore	**3**	**6**
	Parulidae	Yellow-rumped Warbler (*Setophaga coronata*)	Native, Overwintering	Insectivore/Frugivore	**1**	**147**
	Passerellidae	Field Sparrow (*Spizella pusilla*)	Native, Overwintering	Insectivore/ Frugivore	**0**	**1**
		Song Sparrow (*Melospiza melodia*)	Native, Overwintering	Insectivore/Granivore	**0**	**6**
		White-throated Sparrow (*Zonotrichia albicollis*)	Native, Overwintering	Insectivore/Frugivore	**0**	**9**
	Regulidae	Ruby-crowned Kinglet (*Regulus calendula*)	Native, Overwintering	Insectivore/Frugivore	**4**	**11**
	Sturnidae	European Starling (*Sturnus vulgaris*)	Nonnative, Overwintering	Omnivore	**0**	**20**
	Turdidae	American Robin (*Turdus migratorius*)	Native, Migratory	Frugivore/Insectivore	**0**	**5**
		Eastern Bluebird (*Sialia sialis*)	Native, Overwintering	Insectivore/ Frugivore	**0**	**34**
	Tyrannidae	Eastern Phoebe (*Sayornis phoebe*)	Native, Migratory	Insectivore/ Frugivore	**1**	**1**
Piciformes	Picidae	Downy Woodpecker (*Picoides pubescens*)	Native, Overwintering	Insectivore/ Frugivore	**27**	**7**
		Northern Flicker (*Colaptes auratus*)	Native, Overwintering	Insectivore/Frugivore	**4**	**2**
		Pileated Woodpecker (*Dryocopus pileatus*)	Native, Overwintering	Insectivore/Frugivore	**4**	**0**
		Red-bellied Woodpecker (*Melanerpes carolinus*)	Native, Overwintering	Omnivore	**0**	**2**
		Yellow-bellied Sapsucker (*Sphyrapicus varius*)	Native, Overwintering	Insectivore/Frugivore	**1**	**0**

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	Received Marcia Davis Research Award, ETSU Department of Biological Sciences 2018
	Received School of Graduate Studies Research Grant, ETSU 2018
	Received Marcia Davis Memorial Award, Knoxville Chapter of the Tennessee Ornithological Society 2018
	Received 1st Place for Biology Graduate Level Oral Presentations, Appalachian Student Research Forum 2019