Drew University

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White-tailed deer and small mammal population dynamics over time

A Thesis in Environmental Studies and Sustainability (Environmental Science Option)

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ABSTRACT

My research contributed to a long-term study of small mammal and white-tailed deer (*Odocoileus virginianus*) population dynamics over time in Morris County, specifically at Drew University and the Great Swamp Watershed Association. I used live trapping to gather data on small mammal populations and a fecal pellet index (FPI) and visual census to assess the white-tailed deer population. Overall, I was interested in how the population dynamics change over time, independently and interacting with each other. I also started weighing small mammals in an attempt to use weight to clarify small mammal age class.

I found that changes occurred in four target mammal species (raccoon, *Peromyscus* spp., eastern chipmunk, and eastern gray squirrel) over time. Specifically, I noticed an increase in raccoon, squirrel, and *Peromyscus* spp. numbers but a decrease in chipmunk population from 2015-2016. The decrease in chipmunks is a result of older individuals not surviving whereas squirrel numbers increased due to their younger reproducing population. Similarly, the raccoon population had the highest number captured in 2016 compared to previous years, most likely caused by increased reproduction and a greater number of juveniles.

In analyzing relationships between small mammal populations, my study documented a strong relationship between the squirrel and chipmunk populations. I found that over time, chipmunks and squirrels displayed a strong negative relationship in a lagged Pearson correlation and in 2016, I determined that either squirrel or chipmunk activity dominates the trapping grids at Drew University. Moreover, I found that squirrel activity

was significantly higher at the Hepburn Woods site than at the President's House or the Zuck Arboretum sites, when comparing squirrel activity among Drew sites.

My research displayed some interesting findings regarding small mammal weight measurements. On average, Drew University chipmunks and squirrels weighed more than their respective captures at the Great Swamp Watershed Association's sites. I was able to use weight as an indicator of determined age class in chipmunks and squirrels. I found no significant difference in adult weight based on sex. In looking at the impact of deer exclosures on small mammal activity, I found no significant impact from the deer exclosure on small mammal activity.

My study of the white-tailed deer population at Drew University using the fecal pellet index and visual census indicated interesting shifts in the population over time. Of note, there was a marked decline in the population due to epizootic hemorrhagic disease in 2011. However, the population completely recovered in 3-4 years in 2015. By 2016, the population exhibited a crash, most likely due to an overshoot in carrying capacity. The visual census provided a conservative assessment of the age and sex distribution of the deer population. The visual census also indicated that the Drew University population exceeded carrying capacity in 2015, according to Tilghman (1989).

The present findings of this thesis display the importance of long-term research with small mammal and white-tailed deer populations. Future research will continue to be essential in documenting how small mammal and white-tailed deer population dynamics change overtime.

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TABLE OF CONTENTS

I. INTRODUCTION

Overview

I conducted research on small mammal and white-tailed deer population dynamics over time in the summers of 2015 and 2016. My research contributed to a long-term study of mammal populations of Morris County (Drew University and the Great Swamp Watershed Association Conservation Management Area (GSWA-CMA) that was originally designed and implemented in 2009 by Ian DeStefano for his honors thesis project. DeStefano set out to collect baseline data on the small mammal community at Drew University and the GSWA and to analyze how the deer exclosures impacted small mammal populations. Since then, data have been collected annually by Dr. Tammy Windfelder and various research teams every June and July. The small mammal populations were studied using live trapping methods while a fecal pellet index (FPI) and visual census were used to evaluate the white-tailed deer population. My study builds upon previous years' data with the addition of a small mammal weight component and a new site along the Horizon Green Trail at the GSWA-CMA (GSWA-HGT). Broadly, my work with small mammals addresses the following questions:

1. How have four target small mammal populations (raccoon, *Peromyscus* spp., eastern chipmunk, and eastern gray squirrel) changed over time?
2. Do changes in the squirrel and chipmunk populations suggest a competitive relationship?
3. Are there differences in small mammal activity among sites at Drew University?

4. Are there differences in weight of small mammals based on location?
5. Does weight measurement help to clarify small mammal age class?
6. Are there differences in weight of small mammals based on sex?
7. Does the deer exclosure impact small mammal activity?

In measuring the white-tailed deer population using fecal pellet index and visual census data, I developed the following questions:

8. Were there changes in Drew University's white-tailed deer population over time?
9. Were there changes in the population structure between 2015 and 2016 indicated by the visual census?

Before expanding on the details of my study, I will provide a brief review of relevant information regarding the focal small mammal species of my study and then the community as a whole. Following that, the deer population and their relationship with the small mammal community will be reviewed.

Focal Small Mammal Species

Eastern chipmunk (Tamias striatus)

The eastern chipmunk (*Tamias striatus*), a small diurnal ground squirrel, can be identified by the two parallel white stripes that extend from its shoulders down its back (Snyder 1982). The overall body color of the eastern chipmunk varies depending on its geographic location, with the palest colors being closer to Ontario and darker reds in the southern Appalachians (Snyder 1982). New Jersey chipmunks tend to be an intermediate, with a brown coloring and a rusty colored rump. Eastern chipmunks are widely distributed across the eastern United States and southeastern portions of Canada (Snyder 1982; Yahner 1978a). Individual home ranges can vary from 0.01 ha to over a hectare (Snyder 1982). Eastern chipmunk habitat largely consists of deciduous forests, primarily beech maple forests, but can also be found in residential locations that neighbor wooded areas (Mares et al. 1980; Snyder 1982).

Chipmunks have a primarily herbivorous diet of seeds, nuts, and acorns (Snyder 1982; Yerger 1955). However, studies show that *T. striatus* diet fluctuates based on season, age class and sex. Research conducted on the feeding ecology of *T. striatus* found that plants, nuts and acorns make up the majority of their spring, early summer, and fall diet (Svendsen and Wrazen 1978). Most commonly, eastern chipmunk consumed acorns from three species of oak (*Quercus alba*, *Q. rubra* and *Q. velutina*), beechnuts (*Fagus grandifolia*), and hickory nuts (Svendsen and Wrazen 1978). Foraging for fungi and invertebrate material does increase in the summer (Svendsen and Wrazen 1978). Adults were found to consume less invertebrate sources when compared to juveniles (Svendsen and Wrazen 1978). Moreover, adult males ate significantly less invertebrates and more plants than other sex age class combinations (Svendsen and Wrazen 1978). Along with different food preferences, males and females also enter breeding condition at different times.

Males tend to be prepared to breed in early February and reduce their testosterone investment by early August, which can be noticed by shrinking testes (Snyder 1982; Yerger

1955). Females, however, exhibit breeding condition around March when they emerge from torpor (Snyder 1982). Investigations indicate that chipmunk breeding takes place twice a year, in the early spring (late February and March) and in the summer (July) (Yahner and Svendsen 1978). Gestation is typically 31 to 32 days (Snyder 1982). Newborns weigh 2.5 to 5 g and weigh approximately 30 g at 1 month old (Snyder 1982). Juveniles emerge from underground at roughly 7 weeks old (Yahner 1978b). Although Synder (1982) claims that chipmunks reach adult size at 3 months, there are discrepancies on when sexual maturity is reached. Studies report that females can reproduce as early as three to seven months, however, others claim that *T. striatus* are not sexually mature until after their first year (studies reviewed in Yerger 1955).

Weight does appear to differ by age class in *T. striatus*. Because *T. striatus* has two reproductive cycles per year, Pidduck and Falls (1973) looked at weight of juveniles at emergence and found that weight varied from 27.9 to 70.9 g and averaged 50.5 g. Correspondingly, adults were found to weigh more, as Synder (1982) reports that adult weights can vary from less than 80 g to 125 g or more. Average adult weight is 96 g (Tacutu et al. 2013).

Eastern gray squirrel (Sciurus carolinensis)

Eastern gray squirrel (*Sciurus carolinensis*) is a tree squirrel that has three different color morphs, varying from buff, gray, and black (Koprowski 1994). Melanism is more common in the north (Gustafson and VanDruff 1990; Koprowski 1994). In New Jersey, the squirrels tend to be true gray in color and have a white underbelly. They occasionally have a cinnamon color on their haunches (Koprowski 1994; personal observations 2015 and 2016). The gray squirrel is native to the eastern half of the United States and the southern part of Canada (Koprowski 1994). However, they have been introduced in many different states in the United States, as well as in other areas in Canada and Europe (Koprowski 1994).

Gray squirrels tend to occupy hardwood forest habitats that have prevalent understory growth (Brown and Batzli 1984). Gray squirrel diet consists of various nuts, seeds and fruits, particularly acorns, walnuts, beechnuts and hickory seeds (Korschgen 1981). Korschgen (1981) found that in Missouri, gray squirrel food varies seasonally. Year-round they consumed 97 different plant foods and 14 different animal items in total (Korschgen 1981). Squirrels are scatterhoarders and bury nuts < 2 cm below the surface, acorns are preferred when hoarding nuts due to their high fat content (Koprowski 1994). Greater access to food resources can impact home range size (Harestad and Bunnel 1979). Thus, home ranges exhibit great variation, from 0.5 – 20.2 ha, however, they are typically less than 5 ha (Koprowski 1994). Home ranges of male gray squirrels tend to be1.2 times greater than females' home ranges, with natal dispersal being male-biased (Koprowski 1994).

Gray squirrels have two breeding seasons annually (Brown and Yeager 1945). In Illinois, breeding typically occurs in December-February and May-June (Brown and Yeager 1945). Females reach sexual maturity at 5.5 months but typically do not reproduce until 1.25 years or older while males reach sexual maturity later than females, approximately 10-12 months (Koprowski 1994). There is no significant evidence of sexual dimorphism in eastern gray squirrels (Koprowski 1994; Uhlig 1955). Gray squirrels average weight at weaning is 200 g (Tacutu et al. 2013). A study in West Virginia reported similar average weights for adult males (523 g) and females (518 g) (Uhlig 1955). However, adult weight can vary from 300 to 710 g, with an average of 533 g (Koprowski 1994; Tacutu et al. 2013). Squirrel weight also differs by habitat and location, with habitats that contain larger seed crops and locations situated closer to supplemental foods generally correlating with heavier body masses (Reher et al. 2016; Wauters and Dhondt 1989).

***Mouse species (Peromyscus* spp.)**

There are two sympatric species of *Peromyscus* in Morris County, New Jersey: white-footed mouse (*Peromyscus leucopus*) and deer mouse (*P. maniculatus*). *P. leucopus* tend to be brown to gray in coloration with a white underbelly (Lackey et al. 1985). Total length varies between 130 to 205 mm (Lackey et al. 1985). *P. leucopus* tail is 45-100 mm with little distinct delineation and a hind foot length of 17-25 mm (Bunker 2001; Lackey et al. 1985) *P. maniculatus* fur varies in color, from grayish/brown to red fur with a body length that is typically 119-222 mm long (Bunker 2001). The tail length is 45-105 mm with a clear delineation of whiter hairs on the ventral side of the tail (Bunker 2001). *P. maniculatus* hind foot can range from 18-22 mm (Stephens et al. 2014). *P. leucopus* and *P. maniculatus* occupy similar niches as both *P. leucopus* and *P. maniculatus* were more likely to be trapped in forested areas compared to open areas (Stephens et al. 2014).

Peromyscus spp. can be found in a variety of habitats but tend to prefer deciduous woodlands (Lackey et al. 1985; Bunker 2001). *Peromyscus* spp. usage of their habitat indicates that they are as semi-arboreal because of their ability to climb trees (Lackey et al. 1985). Typically, *Peromyscus* spp. are also flexible in their dietary choices given that the have an omnivorous diet (Aguilar 2011; Bunker 2001; Lackey et al. 1985; Wolff et al. 1985). Wolff et al. (1985) compared the seasonal diets of *P. leucopus noveboracensis* and *P. maniculatus nubiterrae* in the Appalachian Mountains in the USA. The authors found that the most common foods were arthropods, fruit, acorns (*Quercus* spp.) and hickory nuts (*Carya* spp.), with no significant difference in overall diet between the two species (Wolff et al. 1985). The diets did vary slightly within seasons, for example, in the summer, *P. leucopus* ate less arthropods than *P. maniculatus,* 44% to 56% respectively, and ate more green vegetation (10.8% compared to 4.7%) (Wolff et al. 1985). Lackey and colleagues (1985) report that *P. leucopus* feeding behavior does tend to be more flexible than *P. maniculatus*. Overall, research indicates that mouse populations increase and decline in a cyclical manner, usually reaching peak inclines or declines at 3 to 4 year intervals (Drost and Fellers 1991; Wendland 1981). *Peromyscus* spp. home ranges average 0.1 ha, though they have been documented as reaching up to 0.6 ha considering resources, population density, and season (Aguilar 2011; Lackey et al. 1985). The largest home ranges are recorded during breeding season and the smallest home ranges are documented during the winter months (Lackey et al. 1985). Male home range tends to be larger than those of the females (Lackey et al. 1985).

Peromyscus spp. exhibit a polygynous mating system (Aguilar 2011; Bunker 200l). Sexual maturity can occur at roughly 44 days of age in males and females, though male sexual maturity has been documented as early as 28 days and female sexual maturity has been documented as late as 73 days (Dice and Bradley 1942; Tacutu et al. 2013). In the northern populations, breeding season usually occurs March – October (Aguilar 2011). Minimum gestation period in non-lactating females is typically 22 days, however, in lactating females, gestation can be extended to roughly 30 days (Bunker 2001; Lackey 1985). Mean litter size varies based on latitude, age, and body size (Lackey 1985). When the litter is born, growth in infant mice occurs quickly. Dice and Bradley (1942) conducted a study measuring growth in *Peromyscus maniculatus*. The authors measured body length, length of tail, hind foot length, ear length and body weight (Dice and Bradley 1942). All seven subspecies of *P. maniculatus* exhibited fast exponential growth in the measured characteristics until it slowed down at roughly six weeks of age, upon reaching subadult age class (Dice and Bradley 1942). Average weight at weaning was 9.2 g in *P. maniculatus* and 9.24 g in *P. leucopus* (Tacutu et al. 2013) and the body weight of new sub adults ranged from 14 -19 g, depending on the subspecies (Dice and Bradley 1942). A study by Stephens et al. (2014), documented higher weights in *P. leucopus* (range: 9-33 g) compared to *P. maniculatus* (range: 7-27 g). Thus, average weight for *P. maniculatus* was 17.4 g and *P. leucopus* was 19.8 g (Stephens et al. 2014). Similarly, based on articles reviewed by Morris (1979), adult weight for *P. maniculatus* ranges from 10-24 g and *P. leucopus* ranges from 12-31 g.

***Raccoon* (*Procyon lotor*)**

Raccoon (*Procyon lotor*) is a mesopredator (mid-sized mammalian carnivore) that is widely distributed across North America, among other continents (Lotze and Anderson

1979; Troyer et al. 2014). They are characterized by their dark facial mask and conspicuously banded tail (Lotze and Anderson 1979). Apart from their two identifying characteristics, raccoons tend to be gray in color, with tones varying based on location (Lotze and Anderson 1979). There is decent variation in total length but males are usually larger than females (males: 634-950 mm, females: 603-909 mm) (Lotze and Anderson 1979). Raccoons can thrive in a variety of habitats and have adjusted well to urbanization. They commonly nest and sleep in hollow trees (Lotze and Anderson 1979).

Raccoons are opportunistic feeders (Hamilton 1936). They are omnivorous and eat a wide range of plant and animal foods (Lotze and Anderson 1979; Parsons et al. 2013). Research conducted by Hamilton (1936) indicated that berries and fruits were most important in raccoon diet, as they occurred in 20 percent of the individuals examined. It is documented that earthworms, crustaceans and insects are the most common animal food sources (Hamilton 1936; Lotze and Anderson 1979). Raccoons are also known to eat the eggs of birds and turtles (Parsons et al. 2013). Home ranges vary seasonally and based on sex (Lotze 1979). Mean home range in radio-collared males is estimated to be 65 ha, while females have a smaller mean home range at 39 ha (Lotze 1979). Male home range expands greatly during mating season (Fox and Dewey 2001).

Mating season extends from late January to August (McKeever 1958; Lotze and Anderson 1979). Raccoons typically have only one litter per year, with litter sizes ranging from 3-7 pups (Fox and Dewey 2001). Females can reach sexual maturity before one year of age while males do not usually breed before their second year, although they have been documented breeding within their first year as well (Fox and Dewey 2001; Lotze and Anderson 1979).

Small mammal communities in North American forests

Community relationships are known to be important in determining mammalian population dynamics and their use of space (da Fonseca and Robinson 1990; Dueser and Shugart 1978). Sympatric species, meaning those that live in the same geographic location, often have different spatial activity cores in order to limit competition (Armitage and Harris 1982). Six sympatric small mammals (*Peromyscus leucopus, P. maniculatus, Microtus pennsylvanicus, Tamias striatus, Zapus hudsonius*, *and Blarina brevicauda*) in Southwestern Ontario were studied to analyze their distribution across habitats (Morris 1979). All species pairs, apart from *Blarina-Zapus*, varied significantly in their habitat utilization (Morris 1979). Morris (1979) relates this to different microhabitat requirements and niche differentiation of the small mammals. *Blarina brevicauda* and *Zapus hudsonius* are different in their diet and living conditions, thus, they do not differ in their microhabitat usage because they are less likely to compete for resources and space (Morris 1979).

North American forests are populated by numerous mammals that occupy similar habitats. Forests tend to have a prevalent insectivore and omnivore populations, including short-tailed shrew (*Blarina brevicauda*) and raccoon (*Procyon lotor)* (Dueser and Shugart 1978; Lotze and Anderson 1979). Small granivorous mammals are common, including eastern chipmunk (*Tamias striatus*), eastern gray squirrel (*Sciurus carolinensis*), white-footed mouse (*Peromyscus leucopus*), and deer mouse (*Peromyscus maniculatus*) (Ivan and Swihart 2000; Nupp and Swihart 2001). Granivorous mammals, such as those mentioned, display evidence of competition (Nupp and Swihart 2001). A study found significant competition between eastern chipmunk and white-footed mouse, with chipmunks negatively impacting the mouse population (Nupp and Swihart 2001). Further, Nupp and Swihart (2001) determined that the fox squirrel (*Sciurus niger*) population negatively impacted eastern chipmunk numbers. In contrast, Brunner et al. (2013) did not find significant evidence of competition when conducting an experiment analyzing shifts in population of *Tamias striatus*, *Sciurus carolinensis*, and *Peromyscus* spp. depending on the removal of one of the aforementioned species. The present study addresses similar small mammal community dynamics in a deciduous forest and individual species changes.

My research, in addition to the standard procedure developed by Ian DeStefano in 2009 relating to small mammal community dynamics, includes weighing the small mammals. I added the weight measurement in an effort to provide clarity on ambiguous age classes of individuals that have been encountered in previous years' studies by Dr. Windfelder and the trapping teams. I also added a new site at the GSWA-CMA, the Horizon Green Trail (GSWA-HGT) in attempt to gather information into the dispersal distances of eastern chipmunks and *Peromyscus* spp. found at the GSWA-CMA. I tested the following hypotheses in response to my questions regarding small mammal population dynamics:

1. **Null hypothesis:** There is no change in the four target small mammal populations (raccoon, *Peromyscus* spp., eastern chipmunk, and eastern gray squirrel) over time.

Alternative hypothesis: There are changes in small mammal populations over time.

2. **Null hypothesis:** Changes in the eastern gray squirrel and eastern chipmunk populations do not suggest a competitive relationship.

   Alternative hypothesis: Changes in the eastern gray squirrel and eastern chipmunk populations suggest a competitive relationship (sensu Nupp and Swihart 2001).

3. **Null hypothesis:** Small mammal activity does not differ among the sites at Drew University.

   Alternative hypothesis: Small mammal activity does differ among the sites at Drew University (sensu Dueser and Shugart 1978).

4. **Null hypothesis:** There are no differences in weight of small mammals based on location.

   Alternative hypothesis: There are differences in weight of small mammals based on location (sensu Wauters and Dhondt 1989).

5. **Null hypothesis:** I did not expect to be able to use weight as an indicator of age class in small mammals.

   Alternative hypothesis: Weight can be used as an indicator of age class in small mammals (sensu Pidduck and Falls 1973; Snyder 1982; Tacutu et al. 2013).

6. **Null hypothesis:** There are no differences in weight of small mammals based on sex (sensu Koprowski 1994; Synder 1982).

Alternative hypothesis: There are differences in weight of small mammals based on sex.

7. **Null hypothesis:** The deer exclosure does not impact small mammal activity.

   Alternative hypothesis: The deer exclosure does impact small mammal activity (sensu Byman et al. 2013; McShea 2000).

My study of small mammal and white-tailed deer populations is a continuation of an honors thesis project started by Ian DeStefano (advised by Dr. Tammy Windfelder) in 2009. DeStefano (2010) developed his study on the small mammal population and the white-tailed deer population to attain baseline numbers and information on activity, abundance, and richness. Since Dr. Windfelder and DeStefano began this study in 2009, Dr. Windfelder has collected summer data annually with student teams. These data include live trapping data for small mammals and fecal pellet index (FPI) and visual census on the white-tailed deer population. Considering the installation of larger and more permanent deer exclosures on Drew University's campus in 2011, DeStefano (2010) was also interested in if the deer exclosures impacted the small mammal populations. I am also interested in how the white-tailed deer population has impacted the small mammal populations over time.

I will provide a review of white-tailed deer (*Odocoileus virginianus*) effects on the small mammal community in a deciduous forest. Additionally, I will introduce factors that influence white-tailed deer population change over time. In order to provide adequate background on changes that have occurred within the Drew University white-tailed deer population, I will also review epizootic hemorrhagic disease (EHD) considering an outbreak occurred at Drew University in 2011.

White-tailed deer (*Odocoileus virginianus*) and its impact on the small mammal community

Overpopulation of white-tailed deer (*Odocoileus virginianus*) and excessive browsing affects forest composition and thus, small mammals. White-tailed deer impact small mammals though competition for food resources or by altering understory and forest habitat (Flowerdew and Ellwood 2001). Of note, white-tailed deer are known to consume both the saplings and seeds of three key trees in eastern chipmunk, eastern gray squirrel and *Peromyscus* spp. diets: oak (*Quercus* spp.), beech (*Fagus grandifolia*), and hickory (*Carya* spp.) (Anderson and Strole 1992; Korschgen 1981; Svendsen and Wrazen 1978; Waller and Alverson 1997; Wolff et al. 1985).

McShea (2000) analyzed the interaction between deer herbivory and acorn production in the previous year and found variation in population numbers of small mammals (2000). McShea (2000) captured significantly higher numbers of eastern chipmunk (*Tamias striatus*) and white-footed mouse (*Peromyscus leucopus*) within the exclosure compared to the control (unfence area), following years of lower acorn masting. This is most likely because of the decreased competition with the deer for the food source in these areas (McShea 2000). Similarly, a 10-year study indicated that more white-footed mice (*Peromyscus leucopus*) were captured within deer exclosure grids than in control (unfenced) grids (Byman et al. 2013). Exclosure studies provide important information in identifying how deer alter small mammal population dynamics.

Exclosure Studies

Excessive ungulate herbivory has such a profound impact on the forest ecosystem that deer have been classified as a "keystone species," a species that affects the distribution and abundance of other species or affects community structure (McShea and Rappole 2000). Research conducted by Boerner and Brinkman (1996) cited deer browsing to be more important than environmental gradients or climate factors when determining seed longevity and mortality. Numerous papers have been published and continue to be published on the role of deer herbivory on the ecosystem (Côté et al. 2004; Pendergast et al. 2016; Tilghman 1989). A study by Stoeckeler et al. (1957) looked at northern hardwood growth in Wisconsin, comparing four deer exclosures to control (unfenced) areas. The authors found that all four exclosures had more trees 4.5 feet high or taller than in the control area. One study site showed that after eight years, the exclosure contained 11,234 sugar maple saplings of 4.5 feet or taller and the control location had only 164 saplings (Stoeckeler et al. 1957). Tilghman (1989) conducted similar studies in Pennsylvania and found that areas with lower deer density had tree seedlings that were two times taller than those with high deer density. Additionally, Tilghman (1989) noted that areas with higher deer density had less diversity in tree seedlings. Tilghman (1989) suggests that deer densities should remain below 0.067 deer/ha to avoid serious impacts on forest composition and regeneration.

White-tailed deer effects on herbaceous plants and understory can also be documented, though it takes more time to recognize results (Waller and Alverson 1997). Deer exclosure studies indicate that species diversity of herbaceous plants increase significantly inside the exclosure (Anderson 1994; Pendergast et al. 2016). In 65 years, Heart’s Content forest in Pennsylvania lost 81 percent of its species due to browsing (Rooney 2001). Flowering plants, such as trillium, are especially vulnerable to deer herbivory. Height of white-flowered trillium (*Trillium grandiflorum*) can be used as an indicator of deer browsing affects because the height of the trillium significantly decreased during successive growing seasons where browsing occurred in a study by Anderson (1994). The forb, *Lactuca canadensis* also suffers tremendously from deer browsing, where the number of flower heads per plant was seven times less on deer browsed individuals (Shelton and Inouye 1995). Browsed *L. canadensis* also had a smaller proportion of large fruits (Shelton and Inouye 1995). Overpopulation of deer does not only alter the small mammal community and vegetation composition of the forest, it can also lead to changes within the deer population itself (Deffendall et al. 2011). The changes within the deer population are most pertinent to my questions related to white-tailed deer as a population rather than the species’ impacts on the ecosystem.

Changes in Deer Population Over Time

There are numerous ecological and anthropogenic factors that cause changes in deer populations. Some well explored factors that impact deer population include: predation, intraspecific competition, interspecific competition, degradation of habitat, overhunting, encounters with vehicles, and disease (Barr and Wolverton 2014; Bergman et al. 2015; Deffendall et al. 2011; Jarnemo and Liberg 2005; Shope et al. 1960). The factors most pertinent to this study are those relating to intraspecific competition due to excess herbivory and disease. High population numbers effect the health of the habitat and thus, the health and functioning of the deer population (Bergman et al. 2015).

Deer population, sex ratio and age structure are density dependent because of carrying capacities of their habitat. Because male deer are larger and require more resources, they have a higher mortality rate than those of the females (McCullough 1999). Females, being easier to maintain according to environmental limits and higher mortality in males often contribute to a skewed sex ratio in ungulates (McCullough 1999). In support of this theory, a study determined that female fawns had higher survivorship than male fawns when weight was a covariate (Bartmann et al. 1992). Bartmann et al. (1992) set up 3 different experimental populations of mule deer within deer exclosures to test the importance of density on fawn mortality. Results indicated that fawn survival rates were inversely related to density of the population. Moreover, starvation was the leading cause of death among fawns (Bartmann et al. 1992). Of note, there is also potential for skew of age distribution because adult females had a higher survival rate than fawns (Bartmann et al. 1992).

Similarly, Deffendall et al. (2011) found that intraspecific competition among high densities of deer can lead to an unhealthy population. This study calculated the carrying capacity for white-tailed deer on one of Georgia's coastal island. The authors found that 11.5 deer (0.008 deer/ha according to Tilghman (1989)) could be sustained annually on the available vegetation (Deffendall et al. 2011). Furthermore, they attributed the emaciation and disease that the population had experienced to an overshoot of the carrying capacity (Deffendall et al. 2011). Greater population density also allows for easier spread of disease (Deffendall et al. 2011; Bergman et al. 2015). The Drew University white-tailed deer population contracted epizootic hemorrhagic disease (EHD) in late summer/early fall of 2011.

Epizootic hemorrhagic disease (EHD)

Epizootic hemorrhagic disease (EHD) is a vector-transmitted disease that targets ruminants. It is specifically born from the biting midges of the genus *Culicoides*. White-tailed deer is a common host for EHD, especially EHD serotype 2 (Quist et al. 1997). The disease was first documented with the name epizootic hemorrhagic disease (formerly known as black tongue) in August 1955 in New Jersey and is characterized by its high mortality rate in white-tailed deer (Shope et al. 1960; Stevens et al. 2015). The symptoms exhibited are akin to bluetongue virus in cattle (Stevens et al. 2015). These symptoms include lack of appetite, subcutaneous swelling of the head, bleeding, dehydration, prostration and high body temperatures (Quist et al. 1997; Stevens et al. 2015). Considering the severity of the symptoms, it is unsurprising that epizootic hemorrhagic disease often kills a large population of the deer who become infected. This is exemplified in Roughton's (1975) study, in which 62% of the captive deer were killed due to infection.

The EHD outbreaks are seasonal in white-tailed deer and usually occur from mid-summer to late fall. The duration of the disease is short as it appears to last roughly one month (Gaydos et al. 2004; Roughton 1975). Outbreaks have occurred in various areas across the United States including Kentucky (Roughton 1975), West Virginia (Gaydos et al. 2004) and New Jersey (Shope et al. 1960). The first outbreak in New Jersey described by Shope et al. (1960) occurred in the counties of Morris, western Essex and northern Somerset during 1955. This outbreak killed an estimated amount of 500-700 white-tailed deer (Shope et al. 1960).

In my study of white-tailed deer, I measured the population using a fecal pellet index (FPI) and a visual census. Given the literature review concerning deer population change over time and considering the methodologies that I used to measure the population, the following hypotheses regarding the white-tailed deer (*Odocoileus virginianus*) population were tested:

8. **Null hypothesis:** There were no changes in Drew University's white-tailed deer population over time.

   Alternative hypothesis: There were changes in the Drew University white-tailed deer population over time (Deffendall et al. 2011; Shope et al. 1960).
9. **Null hypothesis:** There were no changes in the population and age distribution in 2015 and 2016 indicated by the visual census.

   Alternative hypothesis: There were changes in the population and age distribution in 2015 and 2016 indicated by the visual census (Bartmann et al. 1992).

II. METHODS

Study areas

We collected data at five sites in New Jersey: 3 sites within Drew University (40˚45’40.12”N, 74˚25’32.07”W), the Great Swamp Watershed Association’s Conservation Management Area (GSWA-CMA) (40˚45’41.79”N, 74˚29’43.87”W) and the Great Swamp Watershed Association’s Horizon Green Trail (GSWA-HGT) (40˚45’46.79”N, 74˚29’23.84”W). Drew University is a 75.27 ha college campus located in Madison, New Jersey. While nearly the entirety of Drew’s campus was sampled by the white-tailed deer fecal pellet index (FPI) and visual census of white-tailed deer, specific portions of the campus’ 20 ha Forest Preserve were used for the small mammal trapping. The trapping sites in the Drew Forest Preserve include the Zuck Arboretum and Hepburn Woods. The third Drew site, President’s House Woods, is in another area of the Drew forest. The two sites managed by the Great Swamp Watershed Association are also used for small mammal trapping data.

Drew University

Drew University is located in the Piedmont physiographic province within the Loantaka terminal moraine from the Wisconsin glaciation (Webb and Kaunzinger 1993). The 3 forested sites, the Zuck Arboretum, President’s House Woods, and Hepburn Woods, can be described as upland forest populated with oak (the majority being *Quercus alba* and *Q. velutina*), beech (*Fagus grandifolia*), sugar maple (*Acer saccharum*), and tulip tree

(*Liriodendron tulipifera*) (Webb and Kaunzinger 1993; Sara Webb, personal communication, 2017).

Drew University installed a deer exclosure around the Zuck Arboretum and a portion of Hepburn Woods to aid in forest regeneration in the summer of 2011. The President's House Woods at Drew University is situated on the southeastern area of campus and is not within a deer exclosure.

Great Swamp Watershed Association's Conservation Management Area (GSWA-CMA) and Horizon Green Trail (GSWA-HGT)

The Great Swamp Watershed Association's study sites, the Conservation Management Area (GSWA-CMA) and Horizon Green Trail (GSWA-HGT) are located in Morristown, New Jersey in Harding Township. The two sites are roughly 305 m apart, and although the GSWA-HGT is technically within the GSWA-CMA, a distinction is made between sites because trapping occurred at different times and the small mammal populations appear to be independent and the habitat is different at these 2 sites. These sites are located on flats formed by glacial Lake Passaic of the Piedmont (Collins and Anderson 1994).

The GSWA-CMA was established in 1996 and contains 21.45 ha of preserved land, with 9.31 ha of the land being enclosed by a deer exclosure in 2005 (Great Swamp Watershed Association 2017). When the small-mammal project was first started by Ian DeStefano, trapping grid 9 in the CMA site was outside of the deer exclosure. The deer exclosure was expanded between summer 2011 and summer 2012 trapping seasons to include all 3 trapping grids at that site. GSWA-CMA can be classified as mostly fresh water swamp with the forested areas composed of mixed hardwoods. The most abundant tree is red maple (*Acer rubrum*), however, there is also slippery elm (*Ulmus rubra*), shagbark hickory (*Carya ovata*), pin oak (*Quercus palustris*), and American beech (*Fagus grandifolia*) in the GSWA-CMA (Collins and Anderson 1994; C. Gentile, unpublished data, 2017; Great Swamp Watershed Association 2017). The site contains vernal pools and wetlands, as well as the Silver Brook, which feeds into the Passiac River (Great Swamp Watershed Association 2017). Additionally, there are two threatened species documented at the CMA, the wood turtle (*Clemmys insculpta*) and swamp pink (*Helonias bullata*) (Great Swamp Watershed Association 2017). In contrast to the swampy area of the GSWA-CMA site, the GSWA-HGT is both deciduous upland forest and wetland (Martin 2015). The trail was opened in 2015 and features over 750 native trees and shrubs that were planted in effort to promote regrowth after deer browsing (Martin 2015).

Data collection

Ian Destefano began this project in 2009. Since then, data have been collected by various different trapping teams annually in June and July. I collected data on trapping teams in 2015 and 2016. Data were originally collected at the Zuck Arboretum and President's House Woods sites, with the Hepburn Woods site introduced in 2011. I introduced the GSWA-HGT site in the 2016 trapping season. Small mammal data were collected using a live trapping method at Drew University sites and at the GSWA-CMA and GSWA-HGT sites.

Every trapping season (June-July), small mammals are trapped at each site for 5 days resulting in 3,600 trapping hours per site. These hours are standard with the exception of the GSWA-HGT (grid 13-15) in 2016 in which we trapped for only 3 days (72 hours, 2,160 trapping hours) due to destruction of grid 14 by a presumed black bear. The numbers displayed in this thesis represent over 100,000 hours of trapping time

Small Mammal Population Census

Trapping Sites

Small mammals were trapped using Tomahawk and Sherman live traps (available for purchase at www.livetrap.com). There were three trapping sites at Drew University: the Zuck Arboretum, Hepburn Woods and President's House Woods. There was also a trapping site at the Great Swamp Watershed Association's Conservation Management Area (GSWA-CMA) as well as the new trapping site added in 2016 at the Great Swamp Watershed Association's Horizon Green Trail (GSWA-HGT). Within each site, we established three trapping grids (30 total traps in each site). Each trapping grid contained 10 traps of varying sizes (Figure 1). The 3x3 trapping grid contained three rigid mouse/vole traps (10x3x3 in. #101), three chipmunk/rat collapsible traps (16x5x5 in. #201), and three squirrel/muskrat traps (19x6x6 in. #202). All traps had 5 m of space between them. Additionally, one raccoon/feral cat trap (32x10x12 in. #207) was placed outside of the grid. An overview of all trapping sites can be found in Figure 2. Trapping grids 1-3 were in Drew University's Zuck Arboretum, with grids 1 and 2 within the deer exclosure and grid 3 located outside the deer exclosure (Table 1; Figure 3). Grid 1 was in an old deer exclosure so it was protected from deer for a longer time than grid 2 and 3. Grids 4-6 were outside of the deer exclosure in the President's House Woods, on the south side of campus (Table 1; Figure 3). Hepburn Woods contained grids 10-12, with grids 10 and 11 located inside of the deer exclosure of the Forest Preserve and grid 12 located just outside of the exclosure (Table 1; Figure 3). The GSWA-CMA contained grids 7-9, all within a deer exclosure (Table 1; Figure 4). Finally, grids 13-15 were established at GSWA-HGT with no deer exclosure present (Table 1; Figure 4).

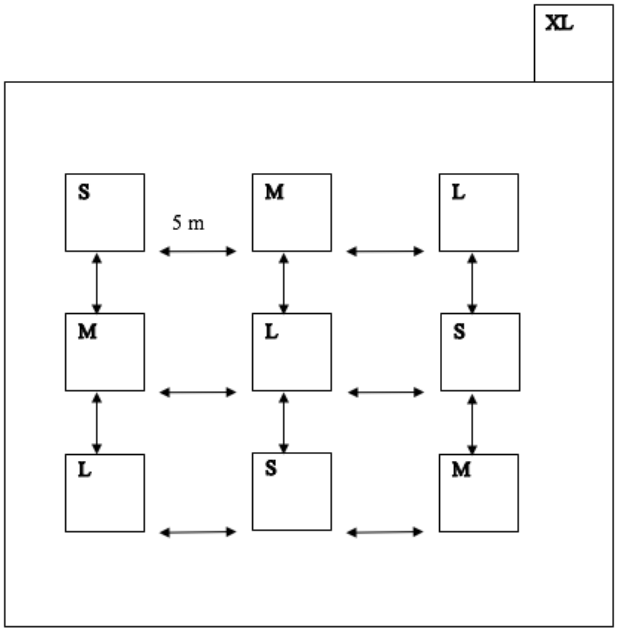


Figure 1. A diagram of a sample 3x3 trapping grid. Each box represents a trap. There is 5 m of space between each trap, represented by the double-sided arrow. S = small (10x3x3 in.), M = medium (16x5x5 in.), L = large (19x6x6 in.), XL = extra-large (32x10x12 in.).

Table 1. Organization of Mammal Trapping Sites

	Site (3 grids/site)	**Trapping Grids Within Deer Exclosure**	**Trapping Grids Outside of Deer Exclosure**	**Sampling Years**
Drew University	Zuck Arboretum	**#1, #2** (note: #1 was in an old deer exclosure -- protected from deer for a longer time)	**#3**	2009-2016
	President's House Woods	All 3 grids are outside the deer exclosure	**#4, #5, #6**	2009-2016
	Hepburn Woods	**#10, #11**	**#12**	2011-2016
Great Swamp Watershed Association (GSWA)	Conservation Management Area (CMA)	**#7, #8, #9**	All 3 grids are inside the deer exclosure (note: #9 was outside deer exclosure at the beginning of this project)	2009-2016
	Horizon Green Trail (HGT)	All 3 grids are outside the deer exclosure	**#13, #14, #15**	2016

Figure 2. A satellite image of the location of the all trapping grids. Shown on the right of the map are Drew University grids: 1-3, 4-6, 10-12. Displayed on the left of the map are the two sites managed by Great Swamp Watershed Association, Great Swamp Watershed Association Conservation Management Area (GSWA-CMA grids: 7-9) and Great Swamp Watershed Association Horizon Green Trail (GSWA-HGT grids 13-15). Mapped using Google Earth, version 7.1.7.2602.

Figure 3. A satellite image of the trapping grids at Drew University (Zuck Arboretum: grids 1 and 2 (inside deer exclosure), 3 (outside deer exclosure). President's House Woods: grids 4-6 (all outside deer exclosure), Hepburn Woods: grids 10 and 11 (inside deer exclosure), 12 (outside deer exclosure). Mapped using Google Earth, version 7.1.7.2602.

Figure 4. A satellite image of the trapping grids at Great Swamp Watershed Association sites (GSWA-CMA: grids 7-9, GSWA-HGT: grids 13-15). Mapped using Google Earth, version 7.1.7.2602.

Trapping Protocol

I received a rabies vaccine prior to the trapping season in case I came in contact with infected individuals. I followed the Guidelines of the American Society of Mammalogists for the Use of Wild Animals in Research (Gannon et al. 2007; Sikes and Gannon 2011). I also followed the Alberta Wildlife Care Committee Class Protocol #007 for Small Mammal Handling and Trapping (Government of Alberta 2005). Drew University's Institutional Animal Care and Use Committee (13-02) approved my study. In 2016, I operated under the following New Jersey Division of Fish and Wildlife permits: SC 2009-01 for scientific collection of game species and permit 2016043 SC for exotic and nongame species. Data collected for exotic and nongame species from previous years operated under New Jersey Division of Fish and Wildlife permit numbers: SC 2015043, SC 2014055, SC 2013050, SC 2012044, SC 2011-117, SC2010-035, SC 2009113.

Traps were checked a minimum of three times a day: once in the early morning, once in the afternoon and once in the evening. However, during periods of heat or high activity at the site, I checked the traps more frequently (every 2-3 hours). If temperatures were extremely high, I checked traps more often. To minimize the risk of heat stroke and ensure animal welfare, traps were closed if the heat index reached 100˚F or above. The traps were bedded with leaf litter to provide the animals with insulation and covered with a fitted piece of cardboard to offer shade and protection from any inclement weather. The amount of bait was prepared based on size of the trap and contained peanut butter, rolled oats, peanuts, and apple slices.

Mammal Processing

Larger mammals, particularly squirrels and raccoons, were marked with metal ear tags (Monel small animal ear tags 1005-1 or 1005-3 from National Band and Tag Co., Newport, KY) and/or Just For Men black beard and moustache dye. I immobilized the animal with a towel in order to apply the ear tag. I did not attempt to pierce an animal's ear if it was caught in a raccoon/feral cat trap (the extra large trap). Smaller mammals were marked with beard and moustache dye. When marking mammals with the dye, each individual of a species was given a unique dye pattern.

Data collected included: date, time processed, trapping grid, trap size and number, species, age class, sex, ear tag/ID number, and weight. Sex and age class were determined through physical observation of the individual. Sex is determined by looking at the animals anogenital region. Adult males have fully descended testicles and adult females typically have visible nipples. Similarly, I look at the anogenital development and body size in order to determine age classes. Younger individual's sex is identified by looking at how large the space is between the genital nub and the anus. If the space is defined (so that testes can develop there), the individual is classified as a male. If there is little space, the individual is determined to be female. I weighed the mammals by transferring the individual to a mesh or nylon bag and weighing them with a 1000g Pesola scale. Animals were released in the same location of their capture.

There are two prevalent mouse species in Morris county: deer mouse (*Peromyscus maniculatus*) and white-footed mouse (*P. leucopus*). In order to aid in distinguishing between these two species, I measured the right hind foot and the tail length. I also noted the fur coloring and color pattern of the tail. Stephens et al. (2014) reports that *Peromyscus maniculatus* had a longer tail, ear and hind foot length than *P. leucopus*. However, *P. maniculatus* tended to weigh less than *P. leucopus* (Stephens et al. 2014). *P. maniculatus* has a distinctly bicolor tail that can be up to 105 mm, individual's hind foot is usually more than 22 mm, and the body coloring tends to be more gray (Bunker 2001). In contrast, *P. leucopus* has a tail length of 65 to 95 mm that is not distinctly bicolored (Aguilar 2011). *P. leucopus* generally has more of a brown body and a hind foot less than 22 mm (Bunker 2001). Due to the difficulty in identifying *Peromyscus leucopus* and *P. maniculatus* from one another and the possibility of hybridization between the two species in this area, I pooled mouse data as *Peromyscus* spp. for this study.

Statistical Analyses

Statistical analyses were conducted through SPSS Statistics 22. I used a 1 sample *t*-test to analyze if the number of raccoons caught during the 2016 trapping season was significantly different from other trapping seasons. Moreover, I used a Chi-square test to evaluate the differences in age distribution between the 2015 and 2016 raccoon populations. I also conducted a Chi-square to analyze fluctuations in mouse population numbers over time. In order to determine population differences in chipmunks at the GSWA-CMA, I conducted a1 sample *t*-test.

I conducted a Chi-square test of independence to test for differences in chipmunk and squirrel age distribution over time. A lagged Pearson's correlation test was employed to analyze if there was a significant relationship between the population numbers of chipmunks and squirrels that we caught over time. For short-term analyses of chipmunk and squirrel relationships in 2016, I assessed activity at Drew sites using a Chi-square goodness of fit test. More broadly, I looked at site usage of chipmunks and squirrels independently as species using multiple Chi-square tests.

Analysis of Variance tests were performed in statistical evaluations when it was appropriate. I conducted a two-way 2 (site) x 2 (sex) ANOVA to look at the difference in weight of chipmunks at Drew University and the Great Swamp sites as a function of location and sex. Similarly, I ran a 1-sample *t*-test to evaluate squirrel weight by location because only one individual was caught at the Great Swamp sites. In order to analyze weight as a factor of age class and sex in chipmunks, I ran a 2 (sex) x 4 (age class) two-way ANOVA. An LSD post-hoc test was used to determine differences in weight per age class. ANOVA was used to conduct tests on mice weight by age, sex and location. Squirrel weight and age class were tested using 2 (sex) x 3 (age class) two-way ANOVA. Further, I analyzed the interaction between age class and sex on squirrel weight using a Bonferroni correction and LSD post-hoc tests.

To evaluate the effect of deer presence on small mammals at Drew University sites, I conducted Chi-square tests to identify if deer influenced raccoon, mice, chipmunk, and squirrel trap visits in 2016. Chi-square tests were also used to evaluate the effect of deer presence on small mammals over time.

White-tailed Deer Population Measures

Fecal Pellet Index (FPI) of White-tailed Deer (*Odocoileus virginianus*)

Deer population data were collected at Drew University and were gathered through fecal pellet index (FPI) and visual censuses. The fecal pellet index was conducted 2009-2016 (except 2010) summers, usually in the beginning of June. Time needed to complete a Drew University FPI depends on the number of pellets encountered, but it takes roughly a week to complete. To determine the changes in the white-tailed deer (*Odocoileus virginianus*) population over time, we used Forsyth's 2005 protocol for Fecal Pellet Index (FPI). I conducted the FPI on Drew University's campus with 30 randomly generated transects spread throughout the study area (Table 2). The transect coordinates have been used since the start of the project in 2009. The direction of sampling of the transect was identified through a randomly generated compass bearing (Forsyth 2005). We navigated to the transect start location using a Garmin GPS 72. Each transect measured 150 meters in length and contained 30 circular (1 meter radius) plots with 5 m between each plot (Forsyth 2005). Pellets were only counted if they were intact, which is defined as no visible loss of material, weathering, mold, or cracks (sensu Forsyth 2005). Individual pellets were counted in pellet groups in which one or more pellets were clustered together. Sampling was not conducted during rain or low light in order to reduce the potential to miss pellets in poor lighting (sensu Forsyth 2005).

Statistical Analyses

I bootstrapped the FPI totals from 2009-2016 (FPI was not conducted in 2010). This allowed us to create a 95% confidence intervals and add error bars for each year. To assess recovery of the deer population after EHD, a 1-sample *t*-test was conducted to test 2015 deer population numbers vs. previous years. Additionally, I ran Wilcoxon signed-rank tests to compare the median total pellets counted in: 1) 2009 vs. 2015 (two peak years) and 2) 2015 and 2016.

Table 2. Randomly generated coordinates and compass bearing used in Fecal Pellet Index

Transect	Latitude	Longitude	Bearing
1	40.76128	-74.42420	348
2	40.76133	-74.42178	328
3	40.76086	-74.42078	90
4	40.76256	-74.42220	197
5	40.76321	-74.42278	331
6	40.76275	-74.42582	209
7	40.76096	-74.42675	193
8	40.76140	-74.42740	209
9	40.75905	-74.42790	37
10	40.75873	-74.42903	171
11	40.75848	-74.42930	245
12	40.76036	-74.43098	255
13	40.76011	-74.43268	229
14	40.76283	-74.42871	154
15	40.76162	-74.43002	117
16	40.76300	-74.43063	341
17	40.76387	-74.43010	330
18	40.75877	-74.42960	289
19	40.75800	-74.43065	97
20	40.75802	-74.43032	270
21	40.75731	-74.42943	269
22	40.75685	-74.42710	38
23	40.75721	-74.42872	200
24	40.75740	-74.42813	239
25	40.75753	-74.42775	208
26	40.75680	-74.42730	134
27	40.75593	-74.42693	178
28	40.75760	-74.42722	182
29	40.75842	-74.42635	130
30	40.75710	-74.42867	173

Visual census of white-tailed deer (*Odocoileus virginianus*)

We walked a 4.52 kilometer route through Drew University's campus to visually assess the white-tailed deer population (Figure 5). Procedure has been modified slightly over time but in 2015, 9 dawn and 9 dusk walks were conducted and in 2016 there were 6 dawn and 6 dusk walks. We recorded all deer that we saw while walking the census route. We took note of how many individuals were present as well as the age class (adult or fawn) and sex of each white-tailed deer. The compass bearing to the direction of where the deer was located was recorded as was the GPS location, using a Garmin GPS 72. Additionally, we measured the distance from the route to the deer using a meter tape and estimated the distance from us to the deer. Each visual census took roughly 90 minutes to complete.

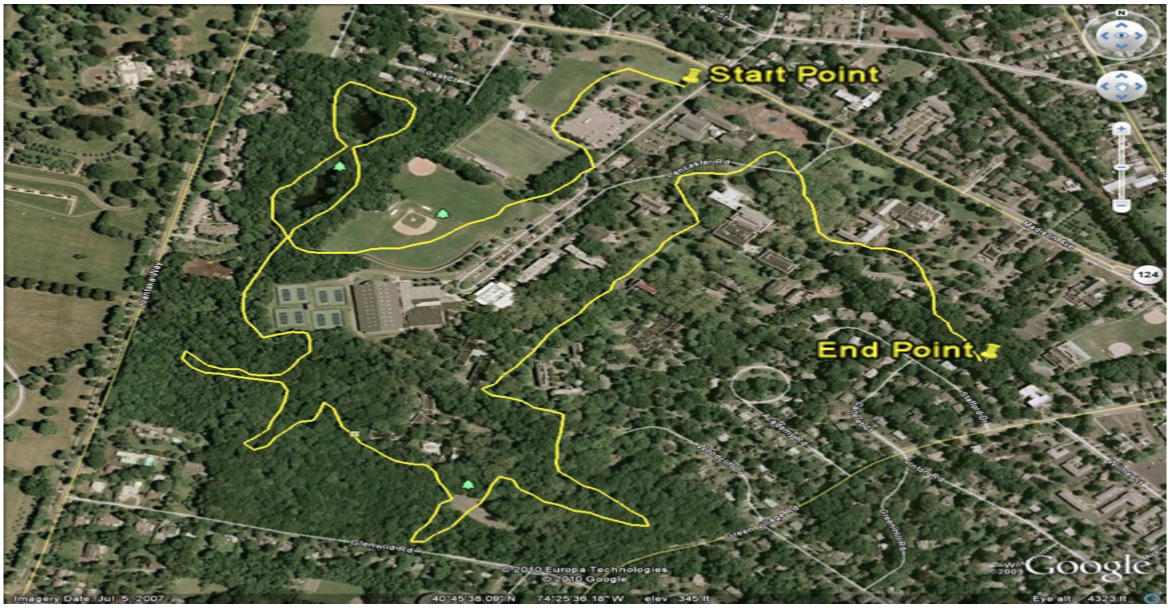


Figure 5. 4.52 kilometer walking route for the visual deer census on Drew University's campus. The path began by the Tilghman lot and ended by the Sitterly lot. Each walk took roughly 90 minutes to complete. Mapped on Google Earth, taken from DeStefano (2010).

III. RESULTS

Small Mammal Live Trapping

Question 1a: Small mammal abundance in 2015 and 2016

Over the course of the 2015 trapping season, the following mammals were captured: eastern chipmunk (*Tamias striatus*) (*N* = 157), eastern gray squirrel (*Sciurus carolinensis*) (*N* = 100), raccoon (*Procyon lotor*) (*N* = 13), mice (*Peromyscus spp.*) (*N* = 11), groundhog (*Marmota monax*) (*N* = 4), Norway rat (*Rattus norvegicus*) (*N* = 1), Southern flying squirrel (*Glaucomys volans*) (*N* = 1). The 2015 numbers can be compared to the 2016 numbers: eastern chipmunk (*Tamias striatus*) (*N* = 94), eastern gray squirrel (*Sciurus carolinensis*) (*N* = 136), raccoon (*Procyon lotor*) (*N* = 30), mice (*Peromyscus maniculatus* and *Peromyscus leucopus*) (*N* = 21), groundhog (*Marmota monax*) (*N* = 2), short-tailed shrew (*Blarina brevicauda*) (*N* =1) and opossum (*Didelphis virginiana*) (*N* = 1) **(**Figure 6**)**. Of the 2016 numbers, 1 squirrel, 7 chipmunks, 3 mice, 1 raccoon, and 1 northern-short tailed shrew were captured at the newly introduced 2016 site, the Great Swamp Association's Horizon Green Trail (GSWA-HGT) (Figure 7).

Excluding GSWA-HGT, there were some notable population shifts. The chipmunk population experienced a 44.6% decrease from 2015 to 2016. More specifically, chipmunk captures at Drew sites (Zuck Arboretum, President's House Woods and Hepburn Woods) decreased from 144 individuals in 2015 to 82 individuals in 2016, a 43% decrease. The squirrel population increased by 35% overall and at Drew sites, seeing as no individuals were capture at the GSWA-CMA (2015: *n* = 100, 2016: *n* = 135). Similarly, the mice (*Peromyscus spp.*) population also exhibited an increase of 63.6% from 2015 to 2016 when evaluating all comparable sites and an 83% increase when looking at Drew sites specifically (2015: $n = 6$, 2016: $n = 11$). Overall, the raccoon population displayed an increase of 123% (excluding captures from GSWA-HGT; Figure 8), with the majority of the total increase occurring at Drew sites (127% increase from 2015 to 2016 at Drew sites) (2015: $n = 11$, 2016: $n = 25$).

Question 1b: Raccoon Population

The raccoon population has varied throughout this long-term project and in the 2016 trapping season, I caught the greatest number of unique individuals thus far ($n = 30$). To test if the number of unique raccoon captures in 2016 differed from the unique captures in previous years, I conducted a 1 sample t-test. The test indicated that the number of unique raccoons caught in 2016 was significantly higher than in other years (*t*-test: $t_4 = -7.84$, $p = 0.001$; Figure 8).

The raccoon population's increase in 2016 occurred in the number of young individuals captured. In 2016, I captured infant/juveniles and juveniles (infant/juvenile: $n = 5$, juvenile: $n = 9$, subadult: $n = 5$, subadult/adult: $n = 2$, adult: $n = 9$), whereas in 2015, I did not capture individuals in young age classes. The age class distribution differed significantly when comparing the raccoon populations of 2015 and 2016 according to a Chi-square test ($X^2_5 = 12.25$, $p = 0.032$; Figure 9). In 2015, I was able to determine 11 out of the 13 captured unique individuals' age classes (juvenile/subadult: $n = 1$, subadult: $n = 6$, subadult/adult: $n = 1$, adult: $n = 3$). There was a slight decrease in subadult individual captures from 2015 to 2016 and a slight increase of subadult/adult captures from 2015 to 2016.

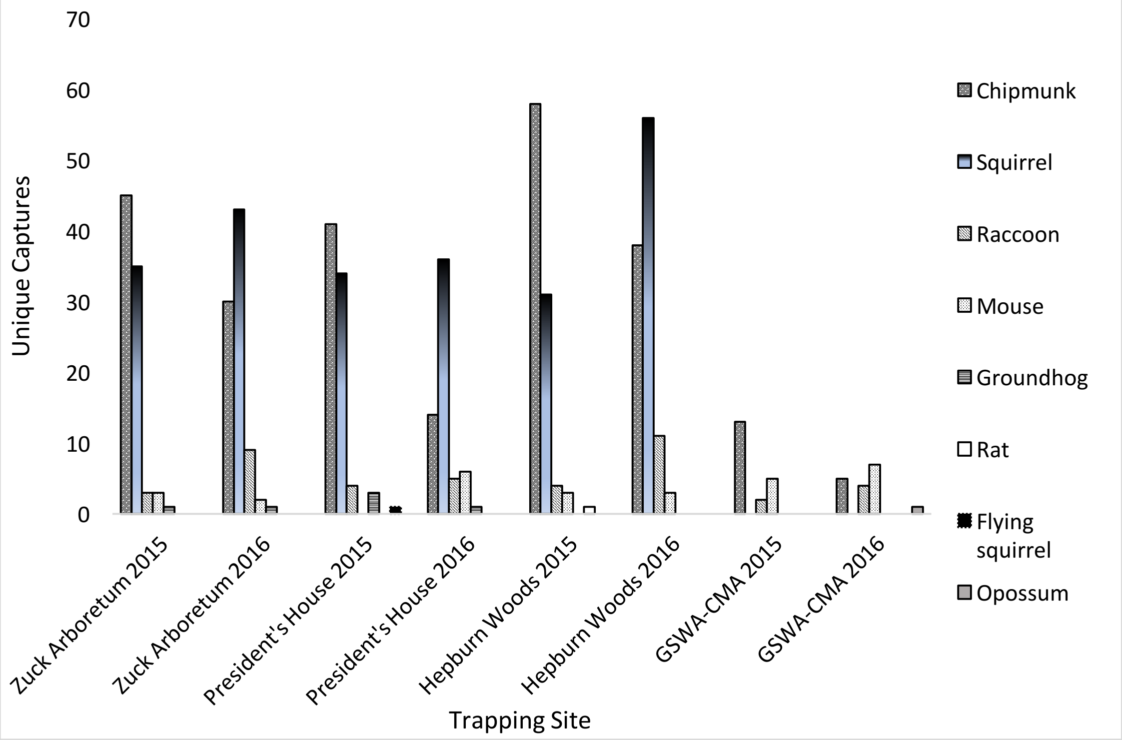


Figure 6. Unique captures during 2015 and 2016 trapping seasons. The Great Swamp Watershed Association's Conservation Management Area is abbreviated as GSWA-CMA. The Great Swamp Watershed Association's Horizon Green Trail (GSWA-HGT) is not included in this graph because it was introduced during the 2016 trapping season.

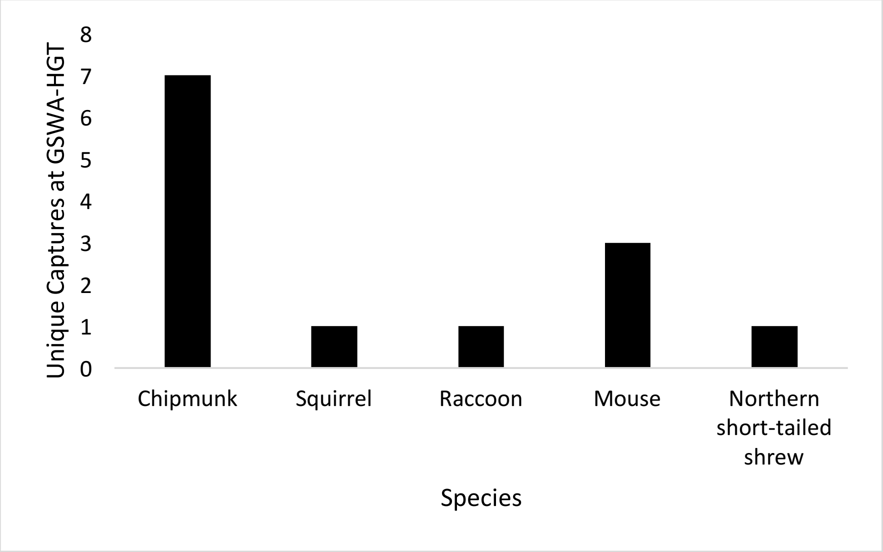


Figure 7. Unique captures at the Great Swamp Association's Horizon Green Trail during 2016 trapping season.

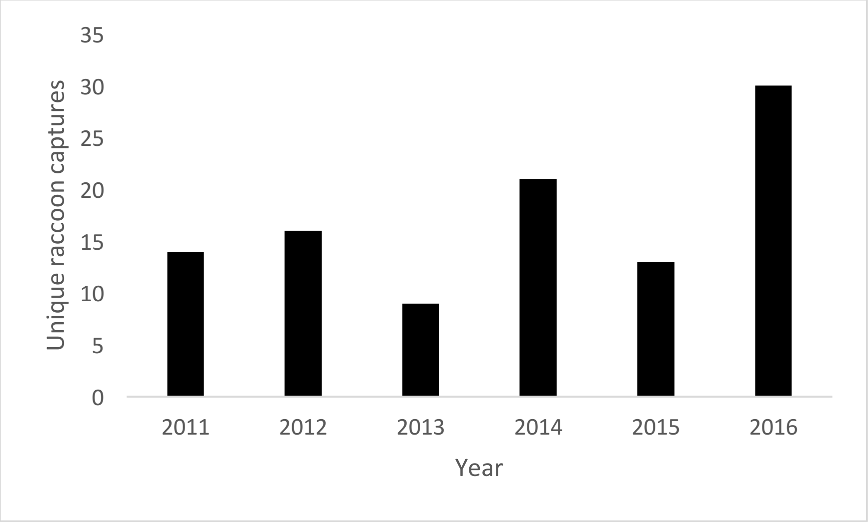


Figure 8. Unique raccoon captures (does not include recaptures) over time. There were statistically significantly more raccoons captured in 2016 (n = 30) compared to other years (2011 (n =14), 2012 (n = 16), 2013 (n = 9), 2014 (n = 21), 2015 (n = 13); (p = 0.001)].

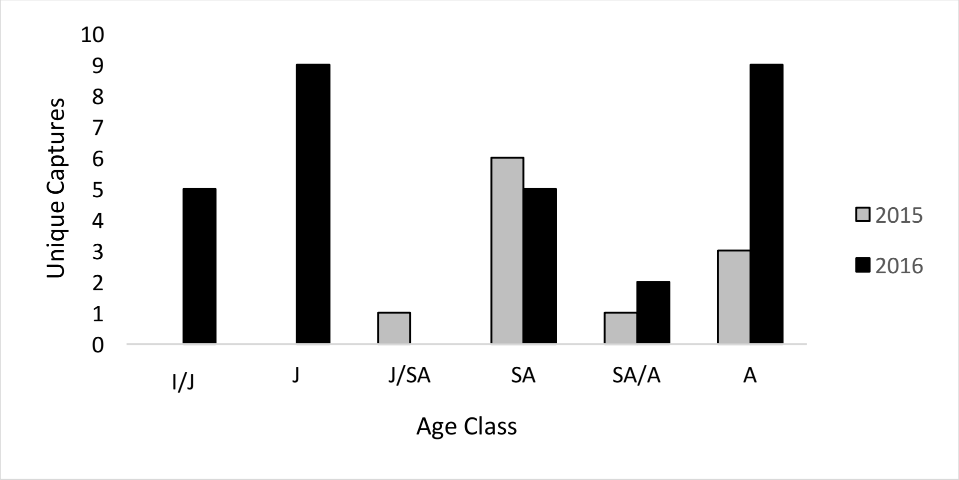


Figure 9. There were significant differences between the age class distributions of the raccoon population in 2015 and 2016. Abbreviations are as follows: I/J: infant/juvenile, J: juvenile, J/SA: juvenile/subadult, SA: subadult, SA/A: subadult/adult, A: adult. 2016: (infant/juvenile: n = 5, juvenile: n = 9, subadult: n = 5, subadult/adult: n = 2, adult: n = 9). 2015: (juvenile/subadult: n = 1, subadult: n = 6, subadult/adult: n = 1, adult: n = 3).

Question 1c: Mice (Peromyscus spp.) Populations

Peromyscus spp. populations also demonstrate fluctuations over time. To determine if the mouse population number varies significantly over time, I conducted a Chi-square test. I found that are significant fluctuations in mouse populations over time ($X^2_7 = 33.39$, $p < 0.001$; Figure 10). I did not include the new GSWA (GSWA-HGT) site in this analysis because the 3 individuals caught at GSWA-HGT would bias the results given the small sample sizes of *Peromyscus* spp. in this study.

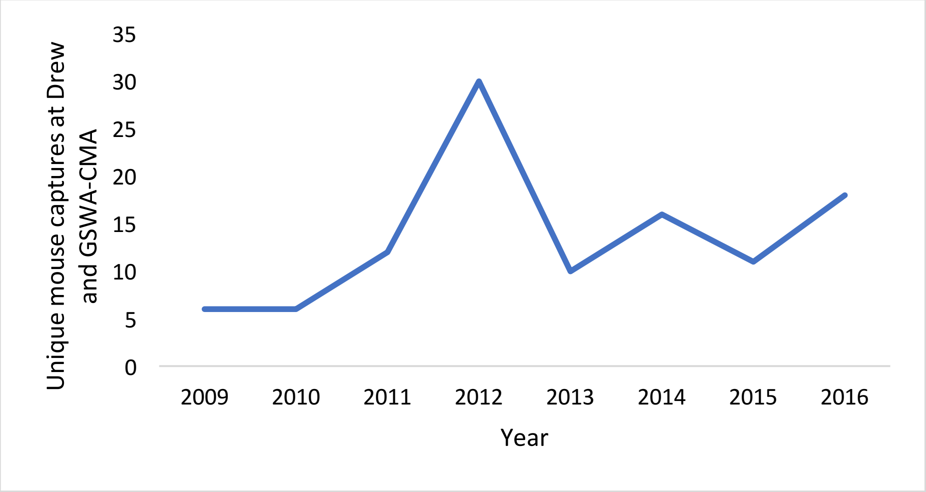


Figure 10. Mouse population over time at Drew and GSWA-CMA. There were significant fluctuations in mouse captures over time (2009 $N = 6$, 2010 $N = 6$, 2011 $N = 12$, 2012 $N = 30$, 2013 $N = 10$, 2014 $N = 16$, 2015 $N = 11$, 2016 $N = 18$).

Question 1d: Chipmunk population dynamics at the Great Swamp Watershed Association's Conservation Management Area (GSWA-CMA) and age distribution over time

During the 2015 trapping season, there was a significant increase in chipmunks captured at the Great Swamp Watershed Association's Conservation Management Area (GSWA-CMA) (n caught in 2015 = 13; t-test: $t_5 = -36.60$, $p < 0.001$; Figure 11). The majority of these chipmunks were subadults ($n = 11$). In the 2016 trapping year, we saw a decrease in chipmunks at the GSWA-CMA ($n = 5$). However, the majority of individuals captured at this site were still subadults ($n = 4$).

Age distribution of the chipmunk population (at all sites) differed significantly over time. The first two years (2009 and 2010) of the study were omitted because Hepburn Woods was not included. We conducted a Chi-square test of independence to determine if age distribution varied over time and we found that the age structure varied significantly over the 6 years (Chi-square test: $X^2_{15} = 27.802$, $p = 0.023$; Figure 12). Moreover, looking at 2015 and 2016 age distribution of the chipmunk population, there were significantly less adults captured in 2016 ($X^2_1 = 13.95$, $p < 0.0005$; Figure 12; Table 3).

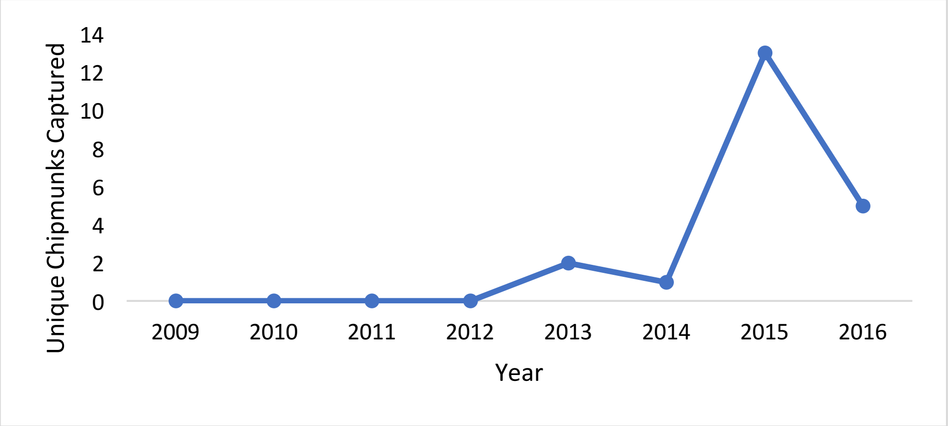


Figure 11. Chipmunk captures at the GSWA-CMA over time (2009: *n* = 0, 2010: *n* = 0, 2011: *n* = 0, 2012: *n* = 0, 2013: *n* = 2, 2014: *n* = 1, 2015: *n* = 13, 2016: *n* = 5). There were statistically significant more chipmunks captured in 2015 compared to previous years (p < 0.0005).

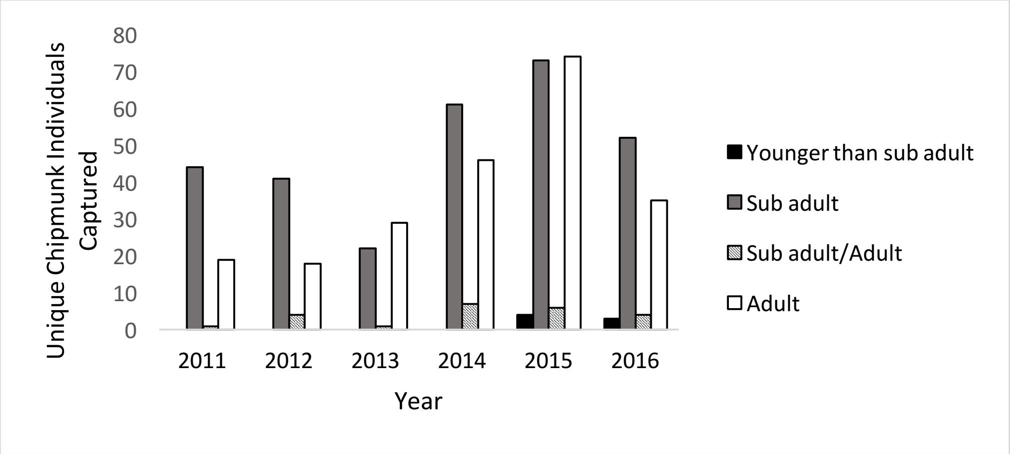


Figure 12. Unique chipmunks caught by year sorted by age class. Chi-square test indicated that chipmunk age distribution varied significantly over time. There were significantly fewer adults captured in 2016 (Table 3).

Table 3. Unique Chipmunks Caught Over Time By Age Class Abbreviations: YSA = younger than subadult, SA = subadult, SA/A = subadult/adult, A = adult

Unique Chipmunks Caught Over Time By Age Class (*n* values)						
	2011	**2012**	**2013**	**2014**	**2015**	**2016**
YSA	0	0	0	0	4	3
SA	44	41	22	61	73	52
SA/A	1	4	1	7	6	4
A	19	18	29	46	74	35
Total (*N*)	64	63	52	114	157	94

Question 1e: Squirrel age distribution over time

Age distribution of the squirrel population also differed significantly over time ($X^2_{15} = 82.65$, $p < 0.0005$; Figure 13; Table 4). Furthermore, there was a significant increase in younger than subadult individuals (juveniles) from 2015 to 2016 ($X^2_1 = 5.4$, $p < 0.05$; Figure 13). Overall, it appears that the squirrel population is reproducing while the chipmunk population is losing older individuals.

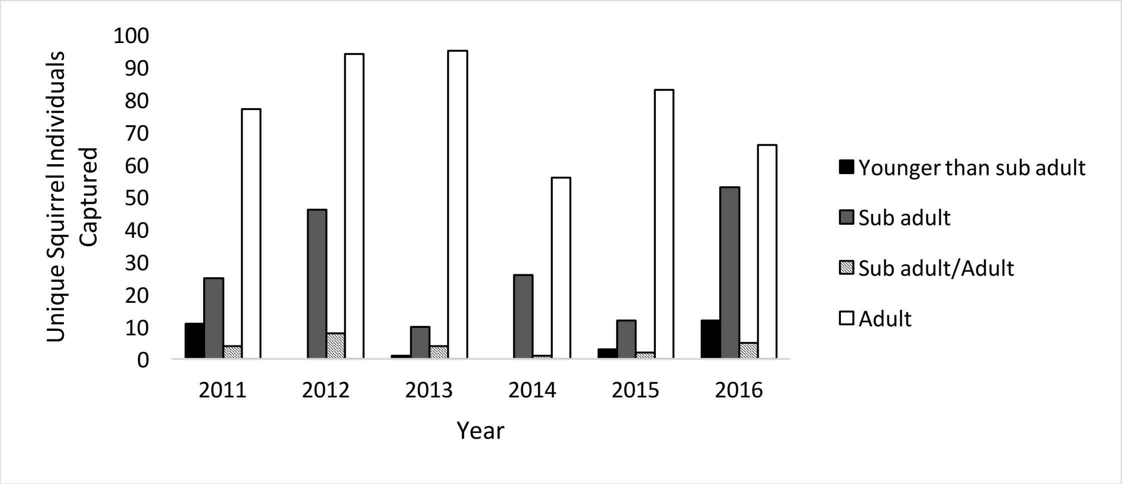


Figure 13. Unique squirrel captures over time grouped by age class. Chi-square test indicated that the squirrel population had significant differences in age distribution over time. There were significantly more younger than subadult individuals captured in 2016 compared to 2015 (Table 4).

Table 4. Unique Squirrels Caught Over Time By Age Class Abbreviations: YSA = younger than subadult, SA = subadult, SA/A = subadult/adult, A = adult

Unique Squirrels Caught Over Time by Age Class (*n* values)						
	2011	**2012**	**2013**	**2014**	**2015**	**2016**
YSA	11	0	1	0	3	12
SA	25	46	10	26	12	53
SA/A	77	8	4	1	2	5
A	19	94	95	56	83	66
Total	132	148	110	83	100	136

Question 2: Inverse relationship between squirrel and chipmunk populations

Eastern chipmunk and eastern gray squirrels are the two most populous species in the study. These two populations have varied over time and thus, I wanted to see if there was a correlation between chipmunk and squirrel population shifts. Analyzing the relationship between squirrel and chipmunk populations over time indicate a strong significant negative correlation between these populations. Years 2011-2016 were included in the analysis because 2009 and 2010 did not include the Hepburn Woods site. I ran a lagged Pearson correlation to look at the relationship of squirrel and chipmunk numbers. Specifically, I lagged the squirrel numbers by one year. For example, 2011 squirrel numbers were matched with 2012 chipmunk numbers and so on. In lagging the data, I found that there is a statistically significant strong negative correlation between the chipmunk and squirrel populations [$r(N = 6$ chipmunk, $N = 5$ squirrel$) = - 0.93$, $p = 0.02$; Figure 14]. This indicates that in years when the squirrel population has high numbers, the chipmunk population numbers the following year are relatively low. Similarly, years with low squirrel numbers correlate with an increase in chipmunk numbers the following year (Figure 14; Table 5).

In addition to the significant long-term inverse relationship between the squirrel and chipmunk populations, I also found significant short term competition on squirrel and chipmunk activity at the site level (Figure 15a-c; Table 6). More specifically, I evaluated if the distribution of squirrel and chipmunk captures are the same across the trapping grids at the level of site in the 2016 trapping season. I conducted a Chi-square goodness of fit test looking at squirrel and chipmunk activity (measured by total captures) distribution at the Drew University trapping sites (Zuck Arboretum, President's House Woods, and Hepburn Woods). There were not enough squirrel captures in the Great Swamp Watershed Association sites to include them. I found that chipmunks have significantly different activity distributions than squirrels across grids in all trapping sites at Drew (Zuck Arboretum Chi-square test: $X^2_2 = 12.35$, $p = 0.003$; President's House Woods Chi-square test: $X^2_2 = 17.38$, $p < 0.0005$; Hepburn Woods Chi-square test: $X^2_2 = 215.83$, $p < 0.0005$; Figures 15a-c).

In the Zuck Arboretum, all three grids were dominated by squirrel activity, although grid 2 was only slightly dominated by squirrel activity in comparison to chipmunk (Figure 15a). The President's House Woods site was also dominated by squirrel activity, however, there was more chipmunk activity in grid 4 compared to grid 5 and 6 (Figure 15b). Hepburn Woods had the most varied activity by chipmunks and squirrels. Notably, grid 10 was dominated by chipmunk activity, while squirrel activity was more prevalent in grids 11 and 12 (Figure 15c).

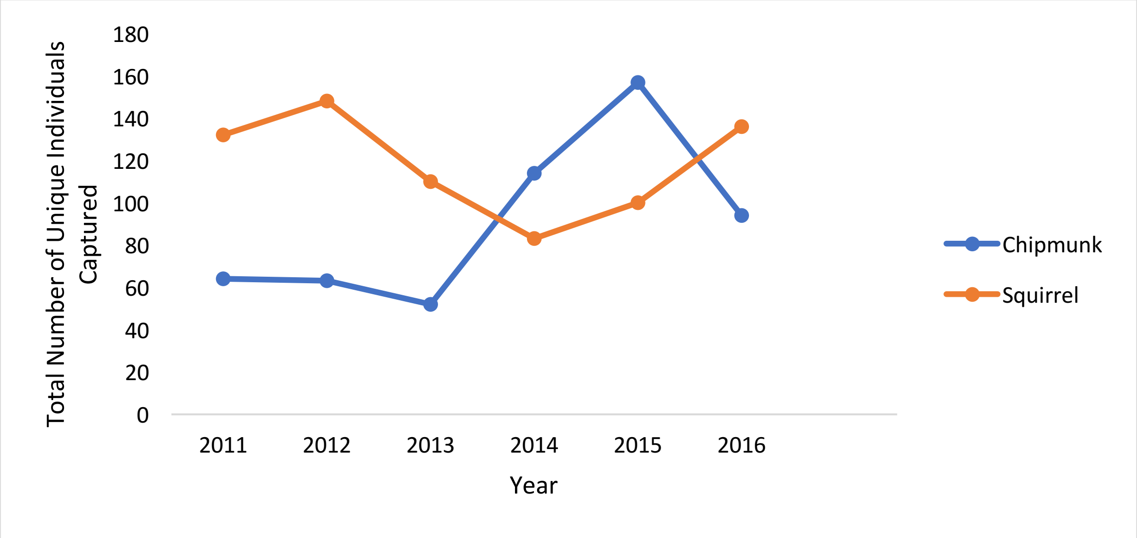


Figure 14. Unique squirrel and chipmunk captures over time. I found a strong statistically significant negative correlation between the squirrel and chipmunk populations. This displays the inverse relationship between the chipmunk and squirrel populations (Table 5).

Table 5. Chipmunk and Squirrel Population Over Time

Year	**Unique Chipmunks Captured (*N*)**	**Total Squirrels Captured (*N*)**
2011	64	132
2012	63	148
2013	52	110
2014	114	83
2015	157	100
2016	94	136

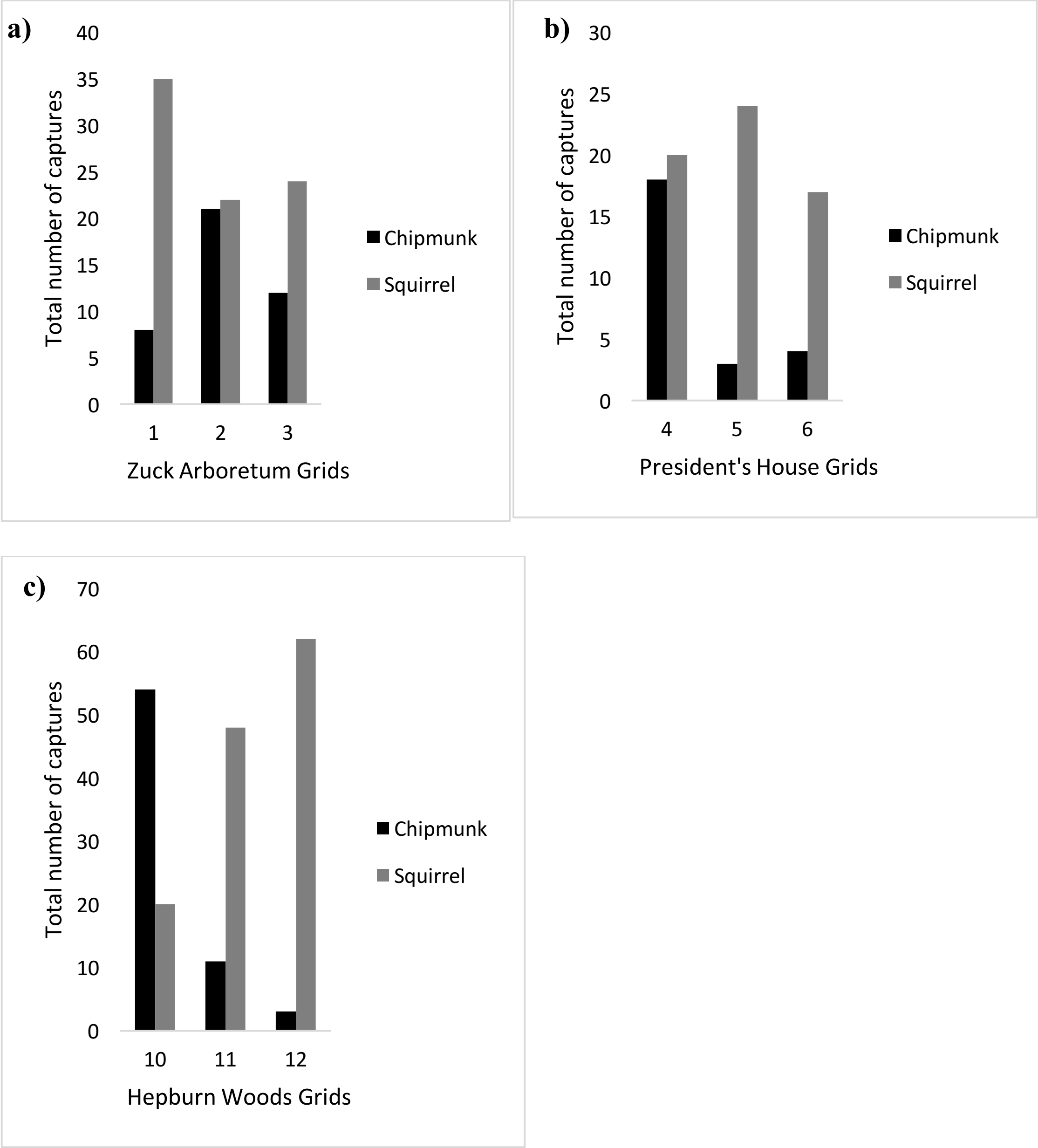


Figure 15a-c. Total number of chipmunk and squirrel captures at Drew trapping sites in 2016. Chipmunks had a significantly different activity distribution than squirrels in all the trapping sites. **a)** Zuck arboretum ($p = 0.003$) (grid 1-3). **b)** President's House Woods ($p < 0.0005$) (grids 4-6). **c)** Hepburn Woods ($p < 0.0005$) (grids 10-12) (Table 6).

Table 6. Total Chipmunk and Squirrel Captures by Site in 2016

Site	Grid	Total Chipmunk Captures in 2016	Total Squirrel Captures in 2016
Zuck Arboretum	1	8	35
	2	21	22
	3	12	24
President's House Woods	4	18	20
	5	3	24
	6	4	17
Hepburn Woods	10	54	20
	11	11	48
	12	3	62

Question 3: Effect of site on small mammal activity

The Chi-square tests evaluating the impact of Drew University site in 2015 and 2016 on activity (measured by total captures) for squirrel was statistically significant. There was significantly more squirrel activity than expected at Hepburn Woods (n = 183) compared to the activity at Zuck Arboretum (n = 135) and President's House Woods (n = 117) during the 2015 and 2016 trapping seasons ($X^2_2 = 16.01$, $p < 0.001$; Figure 16a). There was no significant difference in chipmunk activity among the three Drew sites ($X^2_2 = 5.45$, $p > 0.05$; Figure 16b).

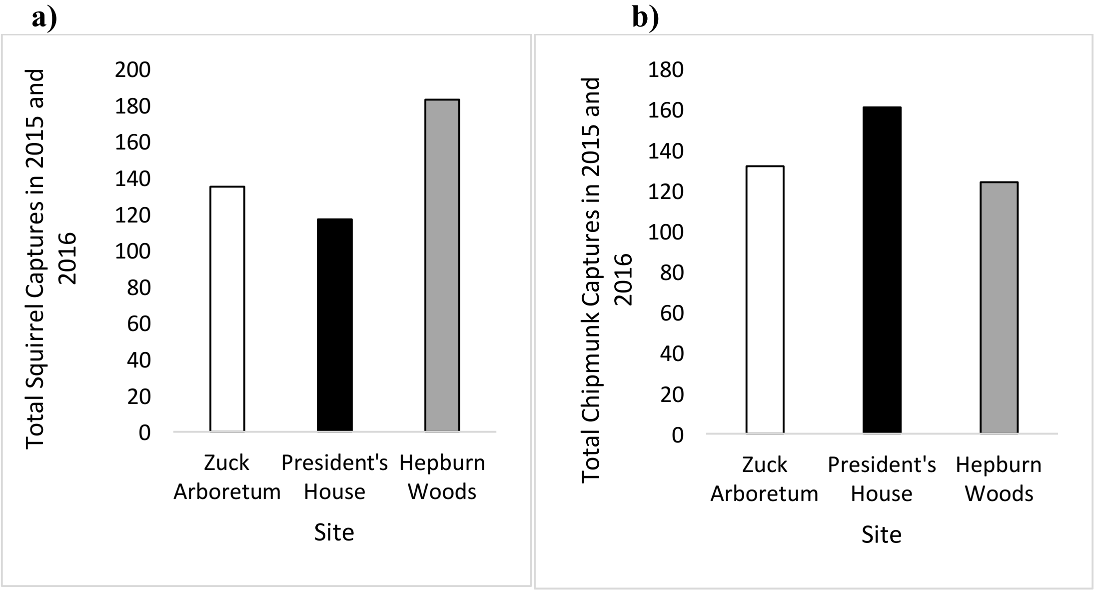


Figure 16a-b. Total captures of squirrels and chipmunks in 2015 and 2016 by Drew University site. There was statistically significantly more activity by squirrels at Hepburn Woods than at Zuck Arboretum and President's House Woods in 2015 and 2016. **a)** Squirrels: Zuck Arboretum (*n* = 135), President's House Woods (*n* = 117), Hepburn Woods (*n* = 183). **b)** Chipmunks: Zuck Arboretum (*n* = 132), President's House Woods (*n* = 161), Hepburn Woods (*n* = 124).

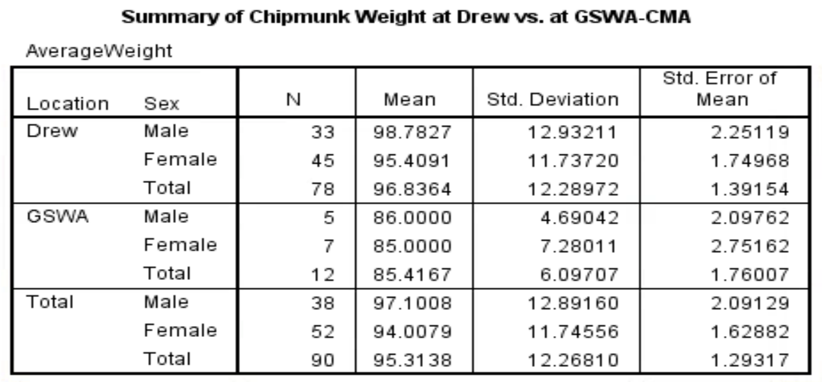
Question 4: Comparing weight at Drew University and the Great Swamp Watershed Association (GSWA)

Chipmunk weight at Drew sites ranged from 73.5 – 121.5 g in males and 75 – 122 g in females. In contrast, weight at the GSWA varied from 80 - 90 g in males and 75 - 95 g in females. I found a significant weight difference in eastern chipmunks by location. In order to determine if weight varied by location and sex for the eastern chipmunk, a 2 (location) x 2 (sex) two-way ANOVA was conducted. There was no statistically significant interaction between sex and location and no statistically significant main effect of sex. However, I did find a statistically significant main effect of location, with chipmunks at Drew weighing significantly more than those at the Great Swamp Watershed Association

(GSWA) on average (ANOVA: $F_{1,86} = 9.90$, $p < 0.0005$; mean Drew University = 97.10 ± 1.34, mean GSWA = 85.50 ± 3.43; Figure 17; Table 7). Drew University chipmunks weighed 12.70 percent more than GSWA chipmunks, on average. R^2 indicated that location is responsible for 10.2 percent of the variance in chipmunk weight. I ran a Chi-square test to ensure that the weight difference was unrelated to age distribution at the different sites. I found that the population structure is not statistically significantly different between Drew University and the GSWA, so age class structure cannot be responsible for the difference in weight ($X^2_3 = 4.32$, $p > 0.05$; Figure 17).

Similarly, I looked at squirrel weight by location. I conducted a 1-sample t-test to compare the weight of subadult squirrels captured at Drew and a subadult individual captured at GSWA-HGT in 2016. I found that, on average, the weight of subadult squirrels at Drew University ($n = 43$, mean = 512.35 g) is significantly more than 435 g, which is the weight of the individual subadult squirrel at the GSWA-HGT (t-test: $t_{42} = 5.93$, $p < 0.0005$; Figure 18). This is a 16.33% percent difference in weight between locations.

Table 7. Average Chipmunk Weight at Drew vs. GSWA-CMA



Summary of Chipmunk Weight at Drew vs. at GSWA-CMA

AverageWeight

Location	Sex	N	Mean	Std. Deviation	Std. Error of Mean
Drew	Male	33	98.7827	12.93211	2.25119
	Female	45	95.4091	11.73720	1.74968
	Total	78	96.8364	12.28972	1.39154
GSWA	Male	5	86.0000	4.69042	2.09762
	Female	7	85.0000	7.28011	2.75162
	Total	12	85.4167	6.09707	1.76007
Total	Male	38	97.1008	12.89160	2.09129
	Female	52	94.0079	11.74556	1.62882
	Total	90	95.3138	12.26810	1.29317

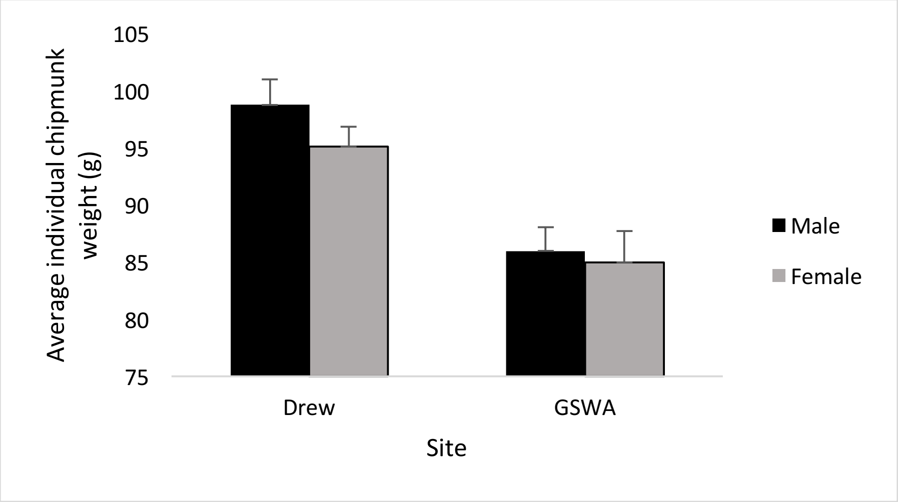


Figure 17. Average individual chipmunk weight by site and sorted by sex. We did not find a significant effect of sex but did determine that Drew chipmunks weigh significantly more than Great Swamp Watershed Association (GSWA) chipmunks. Error bars were calculated using standard error (Table 7)

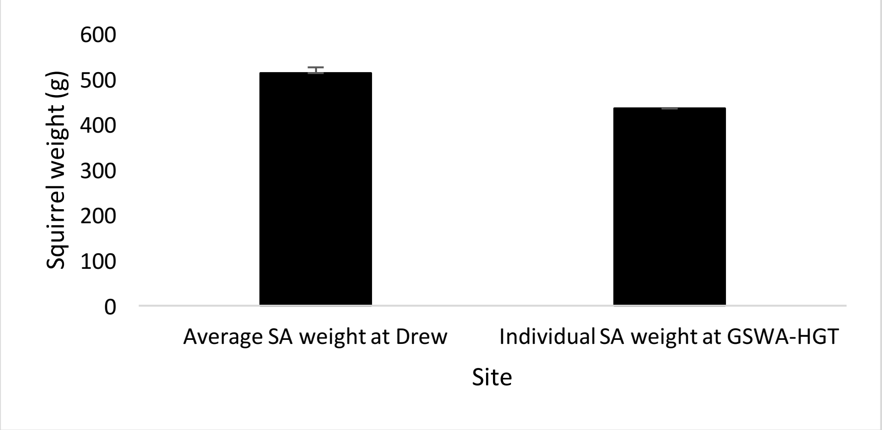


Figure 18. Average subadult (SA) squirrel weight at Drew (*n* = 43, mean = 512.35 g) compared to an individual SA at the Great Swamp Watershed Association's Horizon Green Trail (GSWA-HGT) (*n* = 1 @ 435 g). Error bar calculated using standard error.

Questions 5 and 6a: Weight as a function of age class and sex in the eastern chipmunk (Tamias striatus)

The weight of eastern chipmunks in this study had great variation, ranging from 74 g to 122 g. As expected, the average weight by age class increased with age (Table 8). On average, females tend to weigh slightly less than males in all age categories that were comparable (Table 8). In effort to improve our ability to discern between age classes of small mammals, we evaluated age class and sex to see how those variables relate to individual weight. To determine if average individual chipmunk's weight varies as a function of sex or age class or a combination of both factors, a 2 (sex) x 4 (age class) two-way ANOVA was performed. In order to meet Levene's homogeneity of variances, average individual weight of chipmunk was transformed to log10 for analyses. Normality assumptions were tenable. The two-way ANOVA indicated that there was no statistically significant interaction between sex and age class and no statistically significant main effect of sex. However, there was a statistically significant main effect of age class (ANOVA: $F_{3,83} = 24.03$, $p < 0.0005$; Figure 19a-b). According to R^2, approximately 45.26% of the variance in log10 average weight of individual chipmunk can be explained by age class. Age classes can be defined as younger than subadult, subadult (SA), subadult/adult (SA/A) and adult (A). A LSD post-hoc test displayed statistically significant differences in weight between all age classes except younger than SA vs SA and SA vs SA/A (LSD: $p = 0.02$ for younger than SA vs SA/A, $p < 0.0005$ for younger than SA vs A, $p < 0.0005$ for SA vs A, $p = 0.045$ for SA/A vs A; Figure 19a-b; Table 9). I found no significant main effect of sex on weight (ANOVA: $F_{1,83} = 1.24$, $p > 0.05$; Figure 19a-b; Table 9).

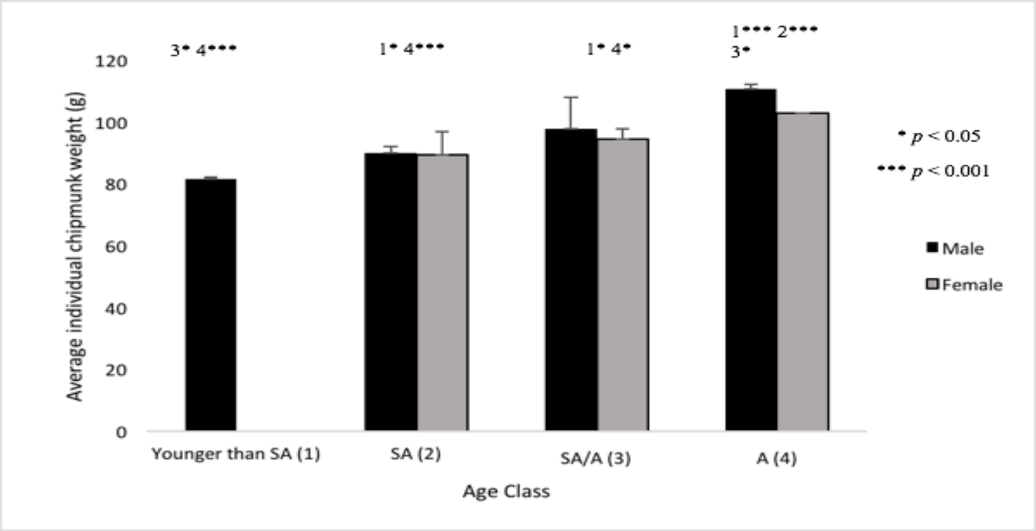


Figure 19a. Average individual chipmunk weight by age class sorted by sex. Younger than SA includes infant, juvenile, and juvenile/subadult. Subadult is abbreviated by SA, subadult/adult is abbreviated by SA/A and adult is abbreviated by A on this graph. 1 = significantly different from Younger than SA, 2 = significantly different from SA, 3 = significantly different from SA/A, 4 = significantly different from A. * = $p < 0.05$, *** = $p < 0.0001$. Error bars were calculated using standard error (Table 8).

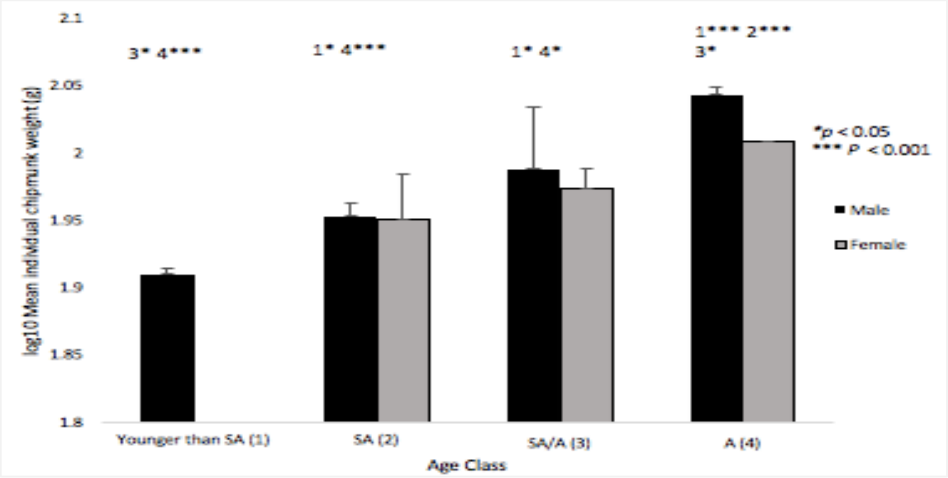
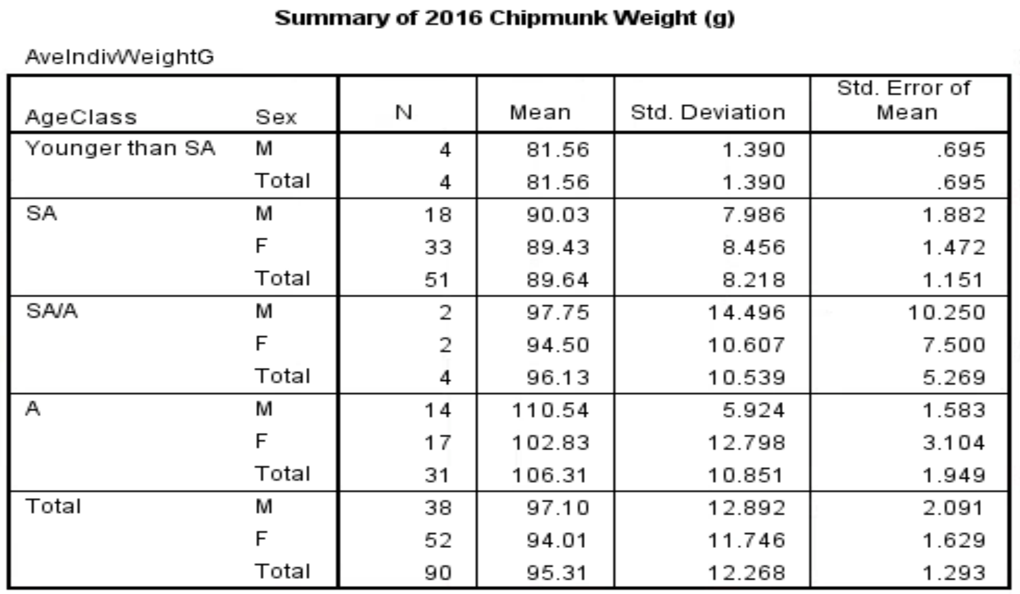


Figure 19b. Log10 transformed values of mean individual chipmunk weight by age class sorted by sex. Younger than SA includes infant, juvenile, and juvenile/subadult. Subadult is abbreviated by SA, subadult/adult is abbreviated by SA/A and adult is abbreviated by A on this graph. 1 = significantly different from Younger than SA, 2 = significantly different from SA, 3 = significantly different from SA/A, 4 = significantly different from A. * = $p < 0.05$, *** = $p < 0.001$. Error bars were calculated using standard error (Table 9).

Table 8. Chipmunk weight (g) in 2016

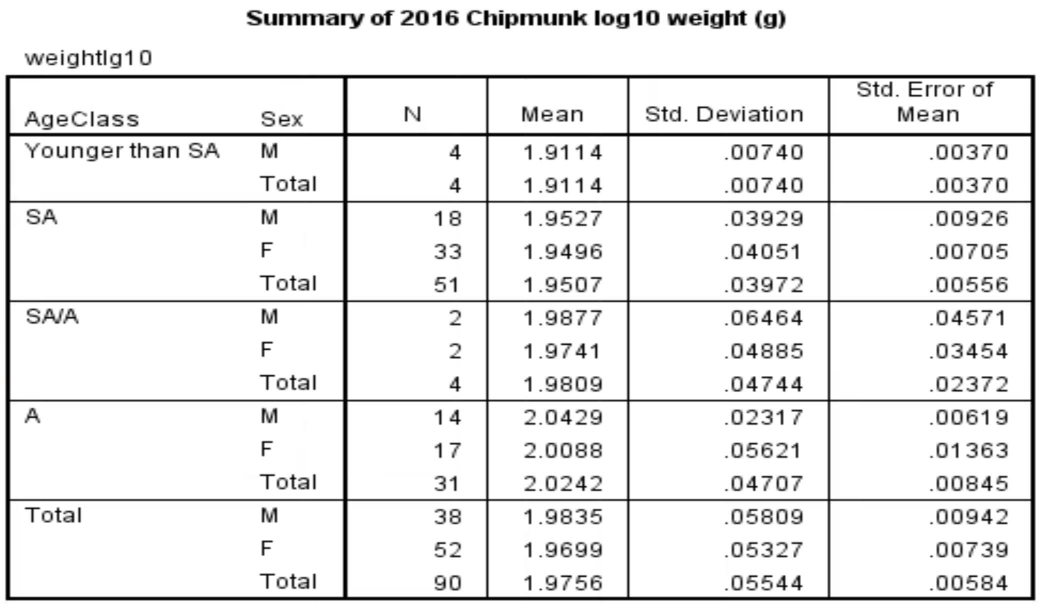


Summary of 2016 Chipmunk Weight (g)

AveIndivWeightG

AgeClass	Sex	N	Mean	Std. Deviation	Std. Error of Mean
Younger than SA	M	4	81.56	1.390	.695
	Total	4	81.56	1.390	.695
SA	M	18	90.03	7.986	1.882
	F	33	89.43	8.456	1.472
	Total	51	89.64	8.218	1.151
SA/A	M	2	97.75	14.496	10.250
	F	2	94.50	10.607	7.500
	Total	4	96.13	10.539	5.269
A	M	14	110.54	5.924	1.583
	F	17	102.83	12.798	3.104
	Total	31	106.31	10.851	1.949
Total	M	38	97.10	12.892	2.091
	F	52	94.01	11.746	1.629
	Total	90	95.31	12.268	1.293

Table 9. log10 transformation of 2016 Chipmunk Weight Abbreviations: SA = subadult, A = adult



Summary of 2016 Chipmunk log10 weight (g)

weightlg10

AgeClass	Sex	N	Mean	Std. Deviation	Std. Error of Mean
Younger than SA	M	4	1.9114	.00740	.00370
	Total	4	1.9114	.00740	.00370
SA	M	18	1.9527	.03929	.00926
	F	33	1.9496	.04051	.00705
	Total	51	1.9507	.03972	.00556
SA/A	M	2	1.9877	.06464	.04571
	F	2	1.9741	.04885	.03454
	Total	4	1.9809	.04744	.02372
A	M	14	2.0429	.02317	.00619
	F	17	2.0088	.05621	.01363
	Total	31	2.0242	.04707	.00845
Total	M	38	1.9835	.05809	.00942
	F	52	1.9699	.05327	.00739
	Total	90	1.9756	.05544	.00584

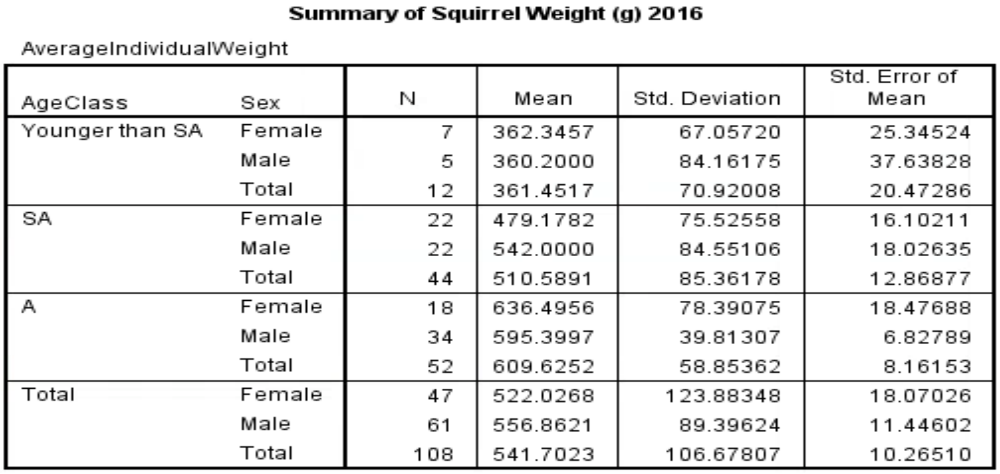
Questions 5 and 6b: Weight as a function of age class and sex in the eastern gray squirrel (Sciurus carolinensis)

The eastern gray squirrel individuals in this study ranged from 255 g to 760 g. On average, younger than subadult males weighed 360.20 g and females weighed 362.35 g (Table 10). Subadult males weighed more than females, on average (Table 10). On average, adult males tended to weigh less than females (Table 10). There was a significant interaction between age class and sex on weight (ANOVA: $F_{2,102} = 6.53$, $p = 0.002$; Figure 20). In analyzing the graph, it appears that younger than subadult weight does not differ between male and females. Subadult males as well as subadult/adult males tend to weigh more than females of the same age class (Figure 20; Figure 21). This relationship changed in adult squirrels; adult females weighed more than adult males, on average (Figure 21). According to R^2, the interaction between sex and age class explains 5.07% of the variance in weight. Subadult/adult individuals ($n = 3$) were left out of the analysis, as the sample size for this age class was too small.

An analysis of simple effects of the interaction of 3 (Age) x 2 (Sex) was conducted using the Bonferroni adjustment to control for family-wise error. Holding sex constant and comparing age determined that all age classes (younger than subadult, subadult, and adult) differed significantly in males and females according to the Bonferroni adjustment ($p = 0.05/2 = 0.025$). Following this result, I conducted LSD post-hoc analyses and found that in both females and males, the adults are the heaviest weight, on average; subadults are heavier than younger than subadults, on average. I found that in holding age constant and comparing sex, only the subadult category had statistically significant mean differences in sex (p = 0.003), according to the Bonferroni correction (p = 0.05/3 = 0.017). However, adult females tended to weigh more than adult males (p = 0.04) The LSD post-hoc analyses of differences in subadults by sex indicated that the subadult weight of females is statistically significantly less than males, on average. I did not test for younger than subadult and adult pairwise comparisons because the univariate tests were not statistically significant.

It is important to acknowledge that in order to run the ANOVA, there are convention assumptions in normality and homogeneity of variance must be met. This ANOVA violates both the normality assumption and homogeneity of variance assumption; however, these violations are minor. Normality is only severely skewed for one cell (subadult male: skewness statistic = -1.071, std. error of skewness = 0.491 → skew = -2.18). Homogeneity of variance is almost met with a p value of 0.041. Despite violating these assumptions, I continued with the ANOVA because the results are valuable given the small p value.

Table 10. Squirrel Weight (g) in 2016 Abbreviations: SA = subadult, A = adult



Summary of Squirrel Weight (g) 2016

AverageIndividualWeight

AgeClass	Sex	N	Mean	Std. Deviation	Std. Error of Mean
Younger than SA	Female	7	362.3457	67.05720	25.34524
	Male	5	360.2000	84.16175	37.63828
	Total	12	361.4517	70.92008	20.47286
SA	Female	22	479.1782	75.52558	16.10211
	Male	22	542.0000	84.55106	18.02635
	Total	44	510.5891	85.36178	12.86877
A	Female	18	636.4956	78.39075	18.47688
	Male	34	595.3997	39.81307	6.82789
	Total	52	609.6252	58.85362	8.16153
Total	Female	47	522.0268	123.88348	18.07026
	Male	61	556.8621	89.39624	11.44602
	Total	108	541.7023	106.67807	10.26510

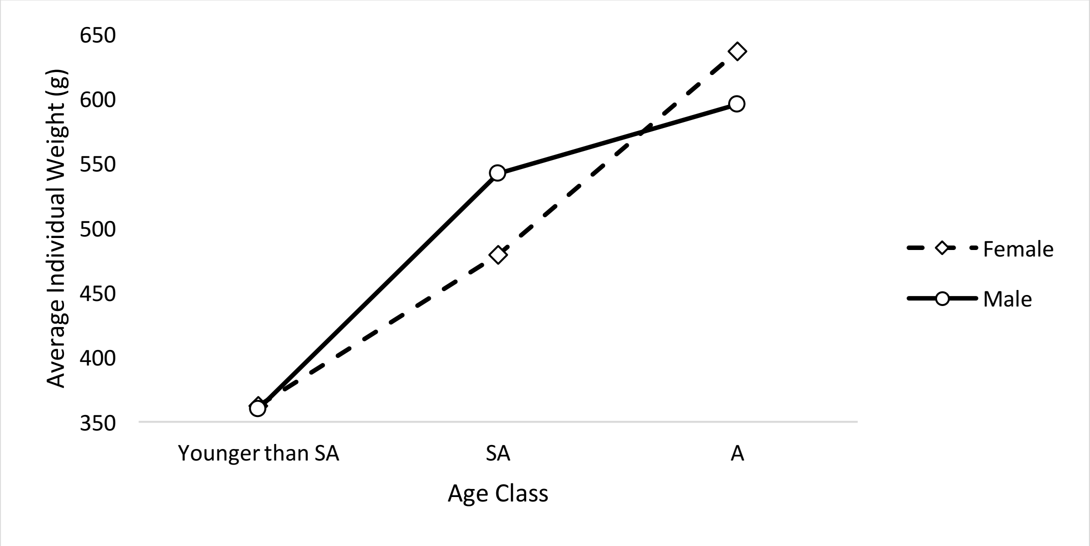


Figure 20. Line graph that displays average individual squirrel weight by age class and sorted by sex. The male trend is depicted using a solid line while the female values is shown in dotted line. Younger than SA includes infant, juvenile, and juvenile/subadult. Subadult is abbreviated by SA, subadult/adult is abbreviated by SA/A and adult is abbreviated by A on this graph (Table 10)

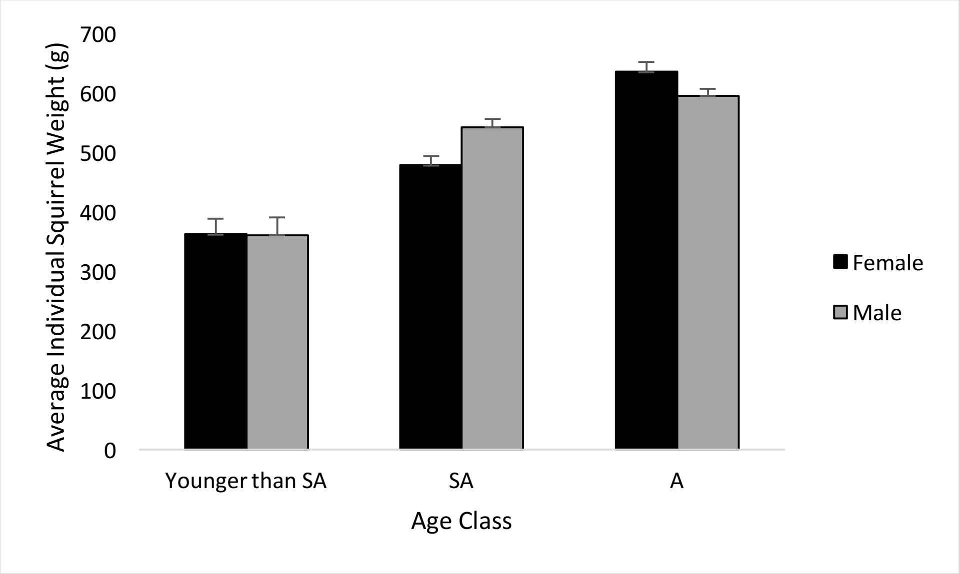


Figure 21. Average squirrel weight by age class and sex. Error bars were calculated using standard error (Table 10).

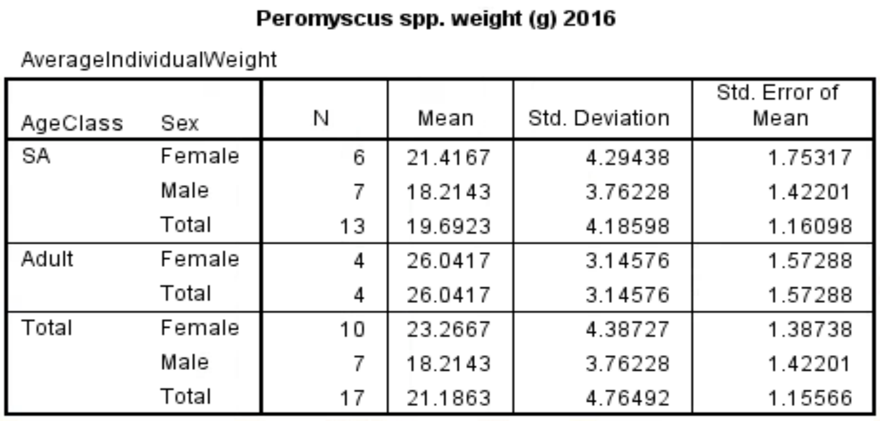
Questions 5 and 6c: Weight as a function of age class and sex in Peromyscus spp.

On average, subadult male *Peromyscus spp.* 18 g and subadult females weighed 22 g (Table 11). Adult females weighed 22 g on average (Table 11). I did not capture any adult males. The minimum weight of *Peromyscus spp.* captured was 22.50 g and the maximum weight was 30 g. Weight was not a useful indicator for age class in mice. In this analysis, I combined the two species of mice that are captured, *Peromyscus maniculatus* and *P. leucopus*, because they are often hard to distinguish and there can be hybridization. I conducted a 2 (sex) x 2 (age class) two-way ANOVA to determine if average individual mouse weight differs as a function of those factors. The interaction could not be evaluated because individuals were not distributed in all categories given the small sample size (n = 17). There was no statistically significant main effect of sex, which indicated that mean weight did not differ significantly between sexes (ANOVA: $F_{1,14} = 2.24$, $p = 0.16$; Figure

22; Table 11). There was also no significant difference in average mouse weight depending on age class (ANOVA: $F_{1,14} = 3.48$, $p = 0.08$; Figure 23; Table 11).

Because I could not properly evaluate the effect of sex on weight given the absence of adult males, I conducted a one-way ANOVA looking at weight as a factor of age class in *Peromyscus spp*. When combining male and female individuals, I found that weight differed significantly by age class, with subadults weighing less than adults, on average (ANOVA: $F_{1,16} = 6.81$, $p = 0.02$; Figure 23a). R^2 indicated that age class is responsible for 30 percent of the variance in mouse weight. In effort to confirm the significant effect of age on weight, I analyzed female mouse weight as a factor of age class by conducting a *t*-test. There was no significant difference between subadult and adult weights in females, on average (*t*-test: $t_8 = -1.59$, $p > 0.05$; Figure 23b). Furthermore, I conducted a *t*-test to look at the effect of sex differences on subadult weight. There was no significant difference in subadult weight in male and females (*t*-test: $t_{11} = -1.47$, $p > 0.05$; Figure 24).

Table 11. Mouse weight (g) in 2016 Abbreviations: SA = subadult



Peromyscus spp. weight (g) 2016

AverageIndividualWeight

AgeClass	Sex	N	Mean	Std. Deviation	Std. Error of Mean
SA	Female	6	21.4167	4.29438	1.75317
	Male	7	18.2143	3.76228	1.42201
	Total	13	19.6923	4.18598	1.16098
Adult	Female	4	26.0417	3.14576	1.57288
	Total	4	26.0417	3.14576	1.57288
Total	Female	10	23.2667	4.38727	1.38738
	Male	7	18.2143	3.76228	1.42201
	Total	17	21.1863	4.76492	1.15566

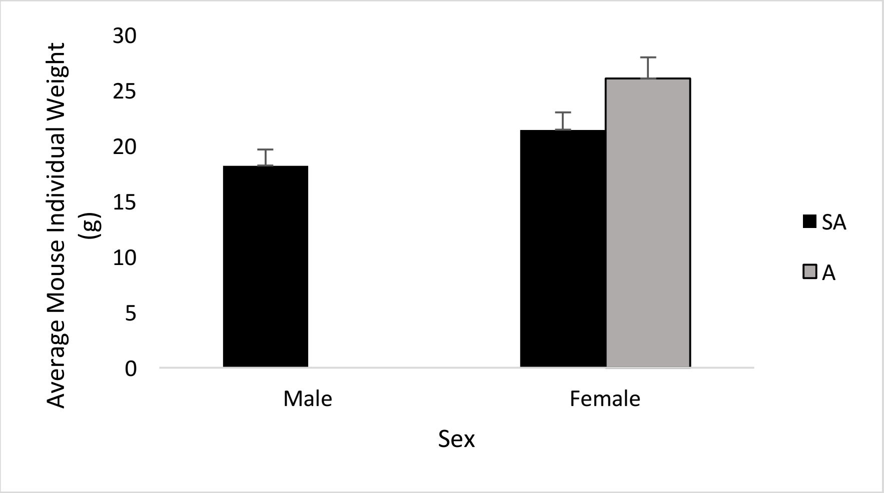


Figure 22. Average individual mouse weight by sex sorted by age class. Subadult is abbreviated by SA (seen in black) and adult is abbreviated by A (seen in gray) on this graph. Error bars were calculated using standard error. We did not find statistically significant impacts of age class or sex on average individual mouse weight.

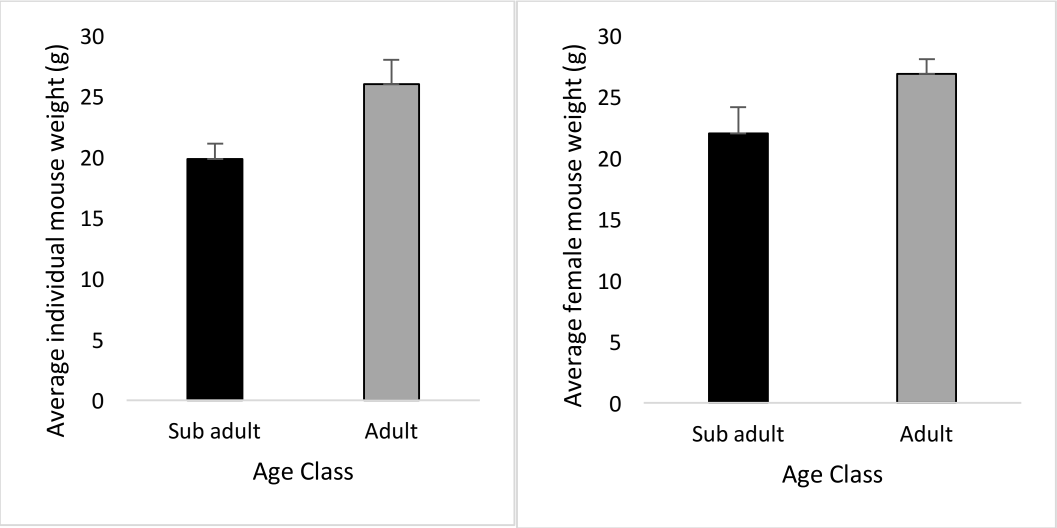


Figure 23. a) Average individual mouse weight by age class. Subadult individuals weighed significantly less than adults, on average ($p = 0.02$). Subadult ($n = 13$), adult ($n = 5$) **b)** Average female mouse weight by age class. There was no significant difference between weight by age class in females. Subadult ($n = 6$), adult ($n = 4$).

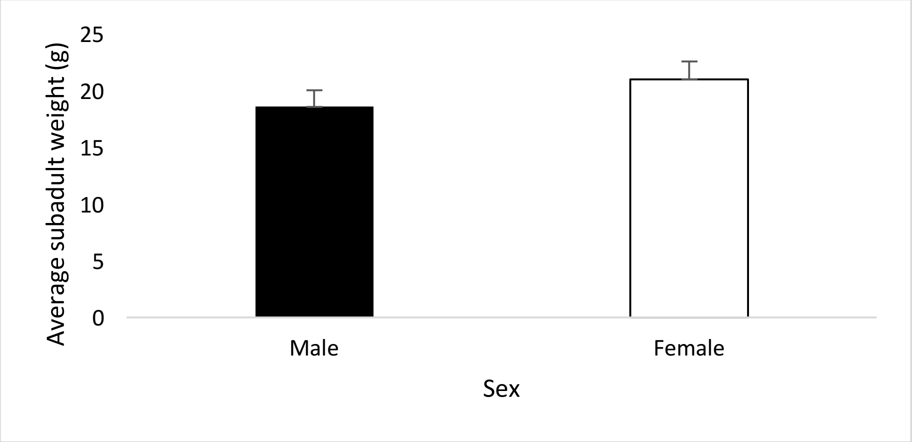


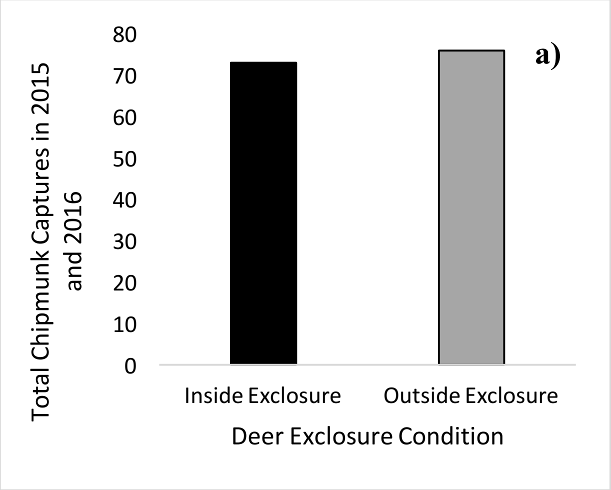
Figure 24. Average subadult weight by sex. I found no significant difference in the weight of male and female subadults. Males (*n* = 7), females (*n* = 6).

Question 7: Effect of deer exclosures on small mammal activity

Deer fencing had no significant influence on small mammal population captures. One of the goals of this long-term project is to investigate the impacts of the deer exclosures on small mammal populations. In order to continue those explorations, I based statistical analyses on pooled capture data from 2015 and 2016. I looked only at the Zuck Arboretum and Hepburn Woods sites because they have grids both inside and outside of the deer fence. This ensured that I could account for any differences occurring at sites, which could impact small mammal activity (measured by total captures of a species). I conducted numerous Chi-square tests by species to identify whether small mammal activity is effected by the presence of the deer fence or a difference in site. I compared grids 2 (inside the deer fence) and 3 (outside of the deer fence) at Zuck Arboretum and grids 11 (within the deer fence) and 12 (outside of the deer fence) at Hepburn Woods.

According to the Chi-square test for chipmunk captures, there was no statistically significant impact of the deer exclosure on chipmunk activity in 2015 and 2016 ($X^2_1 = 0.06$,

$p > 0.05$; Figure 25a). Similarly, I did not find a statistically significant difference in squirrel activity inside vs. outside the deer exclosure in 2015 and 2016 ($X^2_1 = 2.99$, $p > 0.05$; Figure 25b). I ran these Chi-square tests with only squirrels and chipmunks as they are the most populous species at study sites. Considering these did not garner significant results with large sample sizes, we will need a larger sample size on mice in order to evaluate the influence of exclosure on activity.



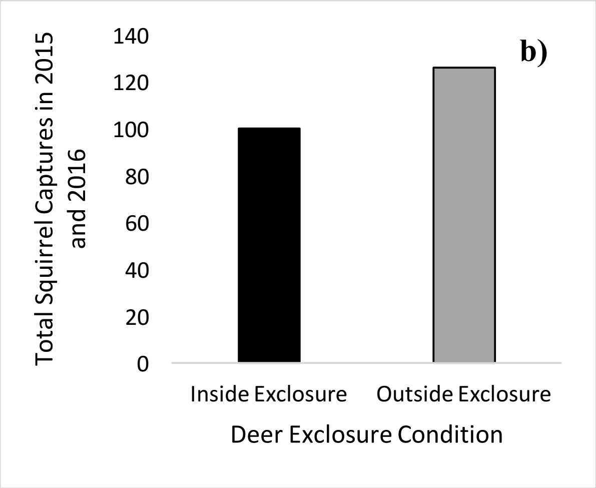


Figure 25a-b. Total captures in 2015 and 2016 by deer exclosure. Values for inside the deer exclosure are combined data from the Zuck Arboretum [grid 2 (inside the exclosure) and Hepburn Woods [grids 11(inside the exclosure)] while values for outside the deer exclosure are combined data from the Zuck Arboretum [grid 3 (outside the exclosure)] and Hepburn Woods site [grid 12 (outside the exclosure)]. **a)** There was no significant difference in chipmunk captures inside (n =73) vs. outside the exclosure (n = 76) ($p > 0.05$). **b)** There was also significant difference in squirrel captures inside (n = 100) vs. outside the exclosure (n = 126) ($p > 0.05$).

Analysis of the white-tailed deer (*Odocoileus virginianus*) population

Question 8: Fecal pellet index (FPI) and Drew University's white-tailed deer (Odocoileus virginianus) population over time

In 2009, we counted a total of 2,600 pellets during fecal pellet index (FPI). FPI was not conducted in 2010. The following FPI in 2011 had only 567 total pellets counted. This represents a 78.2% decrease in population numbers from 2009 (Figure 26). Epizootic hemorrhagic disease (EHD) struck the white-tailed deer population of Morris County in August of 2011 after the 2011 fecal pellet index. There was a decline in total pellets counted from 567 pellets to 170 pellets from 2011-2012. This revealed a significant decrease of white-tailed deer abundance by 70% ($p < 0.001$; Figure 26). However, we counted 2,373 total pellets in 2015. This indicates a statistically significant difference compared to the population numbers of previous years (t-test: $t_4 = -3.46$, $p < 0.026$). In 2016, we counted only 194 fecal pellets, this is a 91.82% decrease from 2015 numbers.

To investigate the difference in total pellets counted in 2009 and 2015, I conducted a Wilcoxon signed-rank test. The median of differences between total pellets counted in 2015 and total pellets counted in 2016 were significantly different (Wilcoxon signed-rank test: $Z = -2.76$, $p = 0.01$; Table 12). 3 transects out of 30 had more pellets in 2016 than in 2015, while 10 transects had less pellets in 2016 than in 2015 (Table 12). The other 17 transects resulted in a tie (Table 12). I also evaluated both population peaks (2009 and 2015) for differences using the Wilcoxon signed-rank test. The median of differences between total pellets in 2009 and total pellets in 2015 was not significantly different (Wilcoxon signed-rank test: $Z = -1.11$, $p = 0.27$; Table 12). 8 transects out of 30 had more pellets in 2015 than in 2009, but 16 transects out of 30 had less pellets in 2015 than 2009. 6 transects had the same pellet counts in 2009 and 2015 (Table 12). Both the 2009 population and the 20015 population were followed by population crashes (Figure 26).

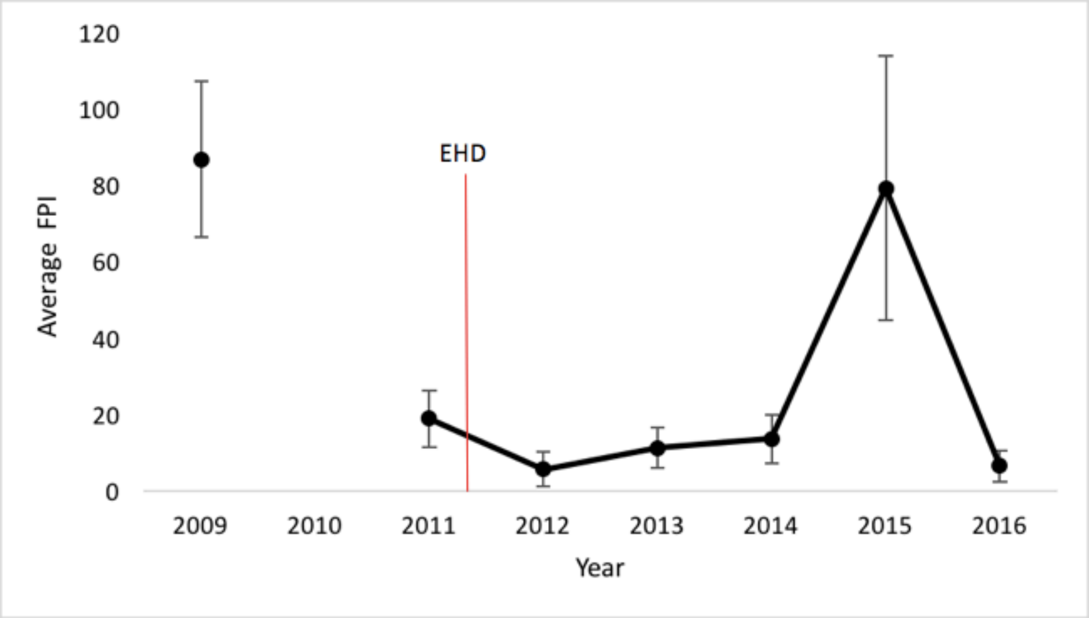


Figure 26. Graph displays the total number of white-tailed deer pellets counted every year when the fecal pellet index (FPI) was conducted. FPI was not conducted in 2010. Error bars were determined using the bootstrap technique. The red line indicates the epizootic hemorrhagic disease (EHD) outbreak.

Table 12. Fecal Pellet Index (FPI) transect totals

Transect	Total Pellets in 2009	Total Pellets in 2015
1	0	0
2	87	0
3	98	0
4	63	0
5	84	0
6	0	0
7	0	0
8	189	0
9	0	0
10	20	32
11	165	39
12	0	110
13	64	0
14	0	0
15	0	0
16	67	0
17	89	0
18	109	255
19	154	290
20	212	32
21	41	0
22	76	18
23	331	0
24	66	0
25	0	195
26	108	974
27	10	17
28	16	111
29	24	0
30	527	300
TOTAL	**2600**	**2373**

Transect	Total Pellets in 2015	Total Pellets in 2016
1	0	0
2	0	0
3	0	0
4	0	0
5	0	0
6	0	0
7	0	0
8	0	0
9	0	0
10	32	35
11	39	0
12	110	0
13	0	0
14	0	0
15	0	0
16	0	0
17	0	0
18	255	0
19	290	0
20	32	0
21	0	0
22	18	0
23	0	0
24	0	14
25	195	117
26	974	0
27	17	23
28	111	0
29	0	0
30	300	5
TOTAL	**2373**	**194**

Question 9: Visual census of white-tailed deer in 2015 and 2016

The maximum number of deer seen on a given census in 2015 was 7 and 4 in 2016 (Table 13). Using the maximum number of deer seen on the visual census, I calculated an estimate of deer/ha, according to Tilghman (1989). In 2015, the visual census indicated that the deer population was 0.093 deer/ha (Figure 27). The 2016 FPI is support by the visual census as it displays that the deer population decreased, with an estimate of 0.053 deer/ha (Figure 27).

I also assessed the age distribution of the deer in 2015 and 2016 using the visual census (Table 14). This was calculated by determining the minimum number of individuals in each age class in each visual census. By looking at the visual census data in this manner, I am able to make conservative estimates of the age composition of Drew University's white-tailed deer population. According to the visual censuses in 2015, the white-tailed deer population had a minimum of 2 adult doe, 1 adult buck and 2 fawns (Table 14). In 2016, the visual censuses indicated that the population had a minimum of 1 adult doe and 3 fawns (Table 14).

In 2015, I counted 24 deer in total during our visual censuses. I detected 20 out of 24 total deer within 25 meters of the trail. This provides a conservative cutoff for a confidence distance at 25 m, as 83.33% of the observations were within 25 m (Figure 28). In contrast, I observed only 16 white-tailed deer throughout the visual censuses in 2016. If using the same confidence distance as calculated for 2015, 43.7% of the deer seen were observed within 25 meters of the trail (Figure 29).

Table 13. Visual Census Observations of White-tailed Deer

Maximum Number of White-tailed Deer Observed on a Visual Census	
Year	**Number of Deer Observed**
2015	7
2016	4

Table 14. Displays the minimum number of white-tailed deer by age class seen on a visual census for both 2015 and 2016. Age class and sex were determined by observation.

White-tailed Deer Age Distribution in 2015 and 2016		
Year	**Age/Sex**	**Minimum # of individuals in each age class (per visual census)**
2015	Adult Doe	4
2015	Adult Buck	1
2015	Fawn	3
2016	Adult Doe	1
2016	Adult Buck	0
2016	Fawn	3

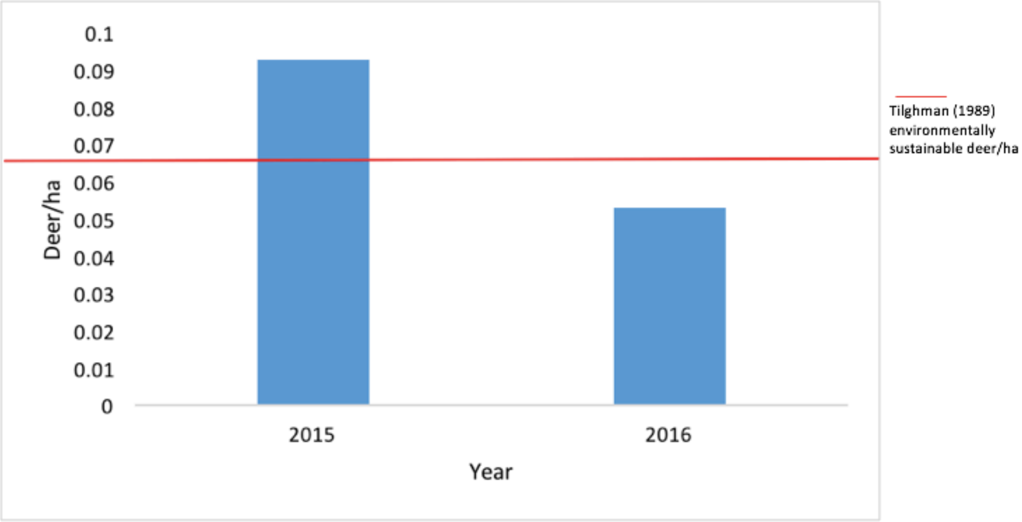


Figure 27. Deer/ha for 2015 (0.093 deer/ha) and 2016 (0.053 deer/ha) calculated using Tilghman (1989) equation. Tilghman (1989) environmentally sustainable number of deer/ha (0.067) is depicted with the red line across the bar graph. 2015 exceeds carrying capacity while 2016 is within carrying capacity.

Figure 28. Histogram displaying number of deer seen in 2015 visual census

Figure 29. Histogram displaying number of deer seen in 2016 visual census

IV. DISCUSSION

Small Mammal Live Trapping

Question 1a: Small mammal population analyses

I attempted to evaluate changes in the small mammal community using long-term data as well as two years of data that I collected myself (summers of 2015 and 2016). In the 2016 trapping season, I added a weight component to examine if weight differs in aspects of age class, sex and location. Moreover, I set out to test if the deer exclosures impact the small mammal population.

In the 2016 trapping season, we caught 7 different species in total: Eastern chipmunk (*Tamias striatus*), eastern gray squirrel (*Sciurus carolinensis*), mice (*Peromyscus spp.*), raccoon (*Procyon lotor*), groundhog (*Marmota monax*), opossum (*Didelphis virginiana*) and short-tailed shrew (*Blarina brevicauda*). With the establishment of the new site, Great Swamp Association's Horizon Green Trail (GSWA-HGT), the northern short-tailed shrew was captured at grid 14. Additionally, 7 unique chipmunks were captured at GSWA-HGT. The 4 remaining sites, excluding GSWA-HGT, displayed a shift in abundance of species.

In evaluating the other four sites (Zuck Arboretum, President's House Woods, Hepburn Woods and GSWA-CMA), I found that small mammal abundance increased for most species in 2016. Excluding captures from GSWA-HGT, eastern gray squirrels (+35%), mice (*Peromyscus spp.*) (+63.6%) and raccoons (+123%) all displayed an increase in their population compared to 2015 numbers. Notably, there was a decline in the eastern chipmunk population (-44.6%) and a significant increase in unique raccoon captures.

Question 1b: Raccoon Population

The raccoon population experienced significant growth, with 2016 captures reaching a record high (*n* = 30). The significant increase in raccoon captures can be attributed to the significant increase in juvenile raccoons in 2016. The juvenile raccoons (*n* = 14) accounted for nearly half of the unique individual raccoons captured (*N* = 30). The juveniles would often be documented in traps with their mother close by or captured in a different trap. The increase in juveniles suggests that more individuals reached sexual maturity and successfully reproduced between the 2015 and 2016 trapping seasons. The subadults (*n* = 6) captured in 2015 may have added to the population. Female raccoons reach sexual maturity at roughly 1 year of age, while males mature later, at 2 years (Kramer et al. 1999). Age and sexual maturity were assessed by external cues, such as size and visible genitals (teats or testes). Thus, there is possibility of incorrect classification between sexually mature and immature individuals. Kramer et al. (1999) incorrectly identified 22.7% (15/66) of females captured as immature using external aging methods. Despite the chance of incorrect age classification of the subadult individuals, it is evident that the population increase in raccoons in 2016 occurred due to a rise in reproduction. It will be interesting to determine if these young individuals will be able to persist and sustain the high raccoon numbers next trapping season.

Question 1c: Mouse (Peromyscus **spp.)** *Population*

I found that *Peromyscus* spp. population numbers fluctuate over time. 2009-2012 displays a strong 4-year cycle. However, the peak following the next four years (2013-2016) was not as strong. Cyclical population change in mice is supported by other research (Drost and Fellers 1991; Wendland 1981). Both Wendland (1981) and Drost and Fellers (1991) documented 3-4 year cyclical peaks or declines in their mouse population studies. An 8-year study by Wendland (1981) credited the marked decline in *Peromyscus maniculatus* to a complete cessation in breeding as well as increased predation by barn owls. It is quite possible that the increase in 2012 is not a part of a cycle, as other factors could be playing a role. Sample sizes for *Peromyscus* spp. are too small to identify if the fluctuation of 2009-2012 was cyclical or due to other factors. Continuing this study will help to determine if the *Peromyscus* spp. population follows a cycle or not.

Question 2: Relationship of eastern chipmunk (Tamias striatus) and eastern gray squirrel (Sciurus carolinenesis)

One of the goals of this project was to evaluate how the chipmunk population at the GSWA-CMA changed following the significant increase of individuals in 2015. In 2015, the majority of chipmunks captured at the GSWA-CMA were classified as subadults (*n* = 11/13). This indicated that individuals were being recruited to the site. However, it appears that the subadult individuals did not colonize or mature at this site. In 2016, I captured roughly half of the number of unique chipmunks (*n* = 5), with 4 of those individuals being subadults. Once again, it appears that a new, albeit smaller, subadult cohort has occupied this site.

Considering the original reason for adding the GSWA-HGT was to evaluate a potential source of chipmunks in 2015, it should be noted that there was no recorded dispersal between the two GSWA sites (GSWA-CMA and GSWA-HGT). Male-biased dispersal between these two sites is possible as males disperse 345 m up to 857 m and females disperse 85 m, on average (Loew 2000; Snyder 1982). It will be interesting to evaluate the population at GSWA-HGT in the future and determine if dispersal does occur between these sites. It will also be important to monitor if the subadults successfully colonize the GSWA-CMA site and how this site's chipmunk population changes over time. The use of long-term data can be a useful indicator in how age distribution fluctuates and changes.

Chipmunk age distribution varied significantly over time. When evaluating age distribution of chipmunks considering the population decline of 2016, I found that significantly fewer adults were captured in 2016 (*n* = 35) than in 2015 (*n* = 74). Given that the eastern chipmunk has a mean life span of 1.3 years but has been documented living up to 8 (Snyder 1982), mortality may be responsible for this noteworthy decrease. Seeing as both 2014 (*n* = 144) and 2015 (*n* = 144) had remarkably high numbers of chipmunks at the Drew sites, it is possible that the population exceeded carrying capacity after two years of sustaining large numbers. In contrast to the decrease in chipmunk adults, there was a significant increase in juvenile (younger than subadult) squirrels. The increase in reproduction in squirrels and the decline in adult chipmunks implicate that there may be a relationship between the two species. Moreover, other results in this study support the hypothesis that the squirrel and chipmunk populations at Drew University are competitive.

Results indicate that the squirrel and chipmunk populations display a significant inverse relationship over time. The strong negative lagged correlation between squirrel and chipmunk populations indicate that when one population experiences growth or decline, the other population exhibits the opposite change. This is especially apparent from the 2015 to 2016 trapping season, as I identified a complete reversal in chipmunk and squirrel abundance. In 2015, the chipmunk population was considerably higher than the squirrel population, however, in 2016, chipmunk numbers fell and the squirrel population increased.

Beyond long-term competition, I found significantly different activity distributions for squirrels and chipmunks among all grids at Drew trapping sites. Almost all grids displayed an inverse relationship between squirrel and chipmunk activity, with either squirrels or chipmunks dominating while the other species total captures remained relatively low. Of note, Zuck Arboretum and President's House Woods grids were largely dominated by squirrel activity. However, Hepburn Woods varied in species dominance, with grid 10 displaying high numbers of chipmunk activity and grids 11and 12 being dominated by squirrel activity.

Competition has been documented between forest rodents, largely squirrels, chipmunks and mice (Nupp and Swihart 2001). Both squirrels and chipmunks consume acorns as a main food staple, which could result in competition over this resource (Koprowski 1994; Snyder 1982). My findings are in contrast to the results of an experimental study conducted by Brunner et al. (2013), in which little evidence for competition was found when analyzing eastern chipmunks (*Tamias striatus*), eastern gray squirrels (*Sciurus carolinensis*), and white-footed mice competition (*Peromyscus leucopus*). Using capture-mark-recapture methods, white-footed mice or gray squirrels were removed from and added to different trapping sites to change the abundance of the species. The authors found inconsistent evidence for competition both in the removal populations and in the reproduction and survival models (Brunner et al. 2013). However, of note, two out of four of the sites with squirrels removed had a significant increase in mice (Brunner et al. 2013). Moreover, the study noted a strongly negative correlation between squirrel and chipmunk numbers in the last month of their study (Brunner et al. 2013). These results indicate that there may be underlying competition that was missed in Brunner et al. (2013) study.

Question 3: The effect of site on small mammal activity

Squirrels had significantly higher activity at Hepburn Woods compared to Zuck Arboretum and President's House Woods in 2015 and 2016. I did not find a significant difference in activity levels in chipmunks by site. A proposed explanation for this heightened activity in Hepburn Woods is that the cicada emergence that occurred in 2013. In 2013, the cicadas were most concentrated in Hepburn Woods. In association with this idea, there is most likely microhabitat differences between sites, which could be a factor in how the small mammals use their space (Ivan and Swihart 2000). I suggest that future research incorporate sampling of the habitats at each site to analyze for microhabitat differences.

An additional layer to this result is predation. It is possible that predation pressure from red-tailed hawk (*Buteo jamaicensis*) may be higher at the Zuck Arboretum. In 2011, there were 7 individuals with puncture wounds/scars caught in the Zuck Arboretum. In 2015, there were 3 unique captures with puncture scars at the Zuck Arboretum. Granted, there is potential for error in these notes, as all scars and puncture wounds may not have been recorded. Regardless, given the higher activity in Hepburn Woods, there is future potential to look at both President's House Woods and Zuck Arboretum as areas with higher giving up densities due to greater perceived predation risk by squirrels (Brown 1999). However, this is under the assumption that the habitats among the Drew University sites are largely similar.

Question 4: Weight at Drew vs Great Swamp Watershed Association's Conservation Management Area (GSWA-CMA) and Horizon Green Trail (GSWA-HGT)

I found that chipmunks at the Drew University sites (Zuck Arboretum, President's House Woods, Hepburn Woods) weigh significantly more than those at the two Great Swamp Watershed Association sites, Conservation Management Area (GSWA-CMA) and Horizon Green Trail (GSWA-HGT). On average, chipmunks weigh 12.7% more at Drew University. This significant difference could not be accounted for by age class distribution in the sample. Similarly, when comparing an individual sub-adult squirrel weight caught at the GSWA-HGT (435 g) to the average weight of subadult squirrels captured at Drew sites (512.35 g), I found that Drew squirrels weigh significantly more than 435 g, on average. It is likely that the populations at the Drew sites have better access to food, given the supplemented food of a college campus, in addition to the surplus of oak trees. Provided food leads to an increase in body mass in small mammals (Banks and Dickman 2000; Reher et al. 2016). Moreover, research indicates that supplemented food causes increases in population numbers due to enhanced immigration (Banks and Dickman 2000). Banks and Dickman (2000) noted that supplemental food increased their wild rodent numbers (*Rattus fuscipes* and *R. lutreolus*) by 4 and 5 fold. Unfortunately, it is hard to track immigration vs. births at the Drew sites but this could be an interesting future development to the project if trapping occurs during multiple seasons.

Habitat differences also offer an explanation for the weight differences between Drew sites and the Great Swamp Watershed Association (GSWA) sites. Wauters and Dhondt (1989) reported a significant difference of weight in red squirrels when comparing two different habitats, coniferous woodland and deciduous woodland. Squirrels in the coniferous woodland weighed more than those in the deciduous habitat (Wauters and Dhondt 1989). The authors credit this marked difference to higher food quality being available and for a longer period of time (Wauters and Dhondt 1989). Although habitat differences at Drew and the GSWA are not as marked as differences in coniferous compared to deciduous woodland, they are notable.

Questions 5 and 6a: Weight as a function of age class and sex in the eastern chipmunk (Tamias striatus)

In attempting to utilize weight as a supportive indicator for ambiguous age class in small mammals, I found a statistically significant effect of age class on average weight in chipmunks. I did not find an interaction between sex and age class on weight nor did I find a significant effect for sex on weight. Weight was indicative of age class in the majority of the categories. Younger than subadult individuals weighed significantly less than subadult/adult individuals and adults, on average. These results are similar to a study by Pidduck and Falls (1973), in which age class distinctions between juveniles and adults were easily made using weight (g). On average, subadults weighed significantly less than adults. Finally, subadult/adults weighed significantly less than adults, on average. However, I found no significant difference between younger than subadult and subadult weight or between subadult weight and subadult/adult weight, on average. Thus, I could not use weight to determine ambiguous age classes.

It appears that weight can be used to support age class predictions that are based on reproductive maturity and size. Weight steadily increased with age, with juveniles exhibiting rapid weight gain (Yahner 1978b). Likewise, a study on yellow-pine chipmunks (*Tamias amoenus*) found that weight gain was fairly consistent during the first 90 days of chipmunk's lives and then leveled off (Broadbooks 1958). Literature also indicated similar weight measurements to those that I attained for adult age class (Pidduck and Falls 1973; Snyder 1982). Considering that Rudd (1955) identified 4 age classes in shrew species, I think it is probable that the four age classes in chipmunks may be distinguished by weight with a larger sample size. Most promisingly, Ford (1981) indicated that weight can be used successfully to determine small mammal's age classes when conducting capture-mark recapture studies. The author reported that with the use of regression models, weight can be just as accurate in predicting age class as other well-known methods (eye lense analyses and skeletal analyses) up to 25 weeks of age (Ford 1981). In support of my results, Ford's (1981) study displayed promise for the use of weight to aid in age class determination of small mammals during field research. However, improvements can be made to enhance age class determination, given the lack of significant differences between younger than subadult vs subadult and subadult vs subadult/adult.

The sample sizes for both younger than subadult age class (*n* = 4) and the subadult/adult age class (*n* = 4) are very small and therefore, may not accurately represent their age class due to large individual variation. A larger sample size is necessary to truly draw conclusions about the insignificant results. There may also be inconsistencies in combining all individuals who are younger than subadult together, as juveniles could be overshadowed by heavier individuals when calculating the average. The smallest younger than subadult chipmunk that I weighed was 81 g while juveniles at five to six weeks weighed 41.7 to 50.0 grams, according to Yerger (1955). However, Tryon and Synder (1973) documented seasonal young as individuals who weighed less than 70 g and Yahner (1978b) caught juveniles at 90 days old that weighed 85 g, thus I may have simply captured late juveniles. A study to determine age in red-tailed chipmunk (*Tamias ruficaudus*) found measurements, including body length and weight to be limited in their power to identify age class (Beg and Hoffmann 1977). This was due to difficulty in identifying age class based on the varied individual means of the measurements following the initial juvenile growth period (Beg and Hoffman 1977). In agreement with this paper, I do not think that weight should be the deciding factor in determining age class for chipmunks. Rather, I believe that weight provides good insight into age class for chipmunks and should be taken into consideration. Given the limitations of live trapping, it is not possible to look at tooth wear, which is largely supported as a reliable measurement of age (Beg and Hoffman 1977; Yerger 1955). Considering this, I recommend that future trapping teams continue weighing small mammals and perhaps adding body length measurement for reinforcement of age class in both chipmunks and squirrels (Beg and Hoffman 1977; Wauters and Dhondt 1989).

Questions 5 and 6b: Weight as a function of age class and sex in the eastern gray squirrel (Sciurus carolinensis)

Similarly, I set out to see if weight could be used to provide insight on squirrel age class. I found an interaction between sex and age class when analyzing weight. In analyzing the interaction and holding sex constant, I found that all age classes differed significantly among males and females (adult > subadult > younger than subadult). In support, Wauters and Dhondt (1989) found that subadults were smaller than adults in all 3 length measurements. However, they did not comment on weight. The only significant difference in weight between the sexes were the subadults. These results are supported by the literature. Females enter estrus at roughly 1.25 years and males reach sexual maturity at 1-2 years (Koprowski 1994). Individuals classified as subadult males based on external signs of sexual maturity could be older and therefore, they would weigh more. Although not statistically significant when using a Bonferroni adjustment, adult females weighed more, on average. I credit this to captures of pregnant individuals who had higher weights and elevated the average (Wauters and Dhondt 1989).

Unfortunately, due to small sample size, I had to leave subadult/adults out of the analysis and combine individuals younger than subadult. In future research, a larger sample size would be beneficial in order to identify if weight can help support the determination of intermediate age classes (infant/juvenile and subadult/adult).

Questions 5 and 6c: Weight as a function of age class and sex in Peromyscus spp.

There was no significant difference in weight in age class or sex in mice. However, I identified some interesting patterns of note. Subadult male *Peromyscus spp.* weighed 18 g and subadult females weighed 22 g, on average. Because no adult males were captured, the two-way ANOVA was difficult to interpret. To analyze the impact of weight on age class, I conducted a one-way ANOVA by lumping the two sexes to look at age class and weight and found that subadults weighed significantly less than adults. This was in the direction that I expected, as adults weighed significantly more than subadults. Given the substantial effect size (30%) of age class on weight, this result is important to note. However, when I evaluated the average weight of subadult females compared to adult females, I did not find significance.

I credit the small sample size for the lack of significant data overall. More mice need to be sampled in order to truly understand if average weight increases significantly by age class. Literature indicates that *Peromyscus leucopus* weigh 12-31 g and *P.*

maniculatus weigh 10-24 g (reviewed by Morris 1979). Although my numbers were within the range of these species weights, the large variation in age class weight may also be impacted by the usage of the 1000 g Pesola scale. Most likely, the 1000 g Pesola was not sensitive enough to measure the weight of the mice. Pizzimenti (1979) weighed small Peruvian rodents with a 100 g Pesola. Thus, I suggest the use of a more exact Pesola to weigh mice in addition to a larger sample size, so that differences in weight by age class can be verified.

Question 7: Effect of deer exclosures on small mammal activity

The long-term data from this study has been useful in trying to determine the impact of deer exclosures on small mammal populations. However, after 5 years of exclosure presence, I have not found any significant effects of the deer exclosure on small mammal activity over time. In contrast, McShea (2000) described differences in small mammal captures within deer exclosures compared to outside deer exclosure. McShea (2000) captured significantly higher numbers of eastern chipmunk (*Tamias striatus*) and white-footed mouse (*Peromyscus leucopus*) within exclosures relative to the control sites following years of lower acorn masting. Most likely, the deer exclosure eliminated small mammal competition with deer for this food source (McShea 2000). McShea (2000) study was conducted for 6-12 years, depending on site. It is quite possible that more long-term data is needed for this study. Pendergast et al. (2016) found little recovery in total forest density and preferred browse density in the first 5 years following deer exclusion at their study site in Pennsylvania. With more time, the authors saw significant species renewal, but not an increase in diversity, over an 11-year period (Pendergast et al. 2016). If my exclosure study site is similar to the one in the aforementioned study, it may need more time to differentiate in plant species renewal and thus impact the small mammal populations.

An issue that is important to consider is that the trapping grids are relatively close together to minimize habitat differences. This allows small mammals to travel between grids easily, despite the presence of the exclosure. I may not be able to detect differences in small mammal population relative to the deer exclosure unless the grids are moved to become more distinct from each other. It will be important to continue to monitor the small mammal population's relationship with the deer exclosure in future summer trapping years.

Conclusions

My study helped to provide valuable results regarding small mammal community dynamics and I was able to answer my questions:

1. I found that changes did occur in the four target small mammal populations over time. Thus, I accepted my alternative hypothesis of: there are changes in small mammal populations over time. Specifically, I found that:
    - In 2016, the chipmunk population decreased from 2015 numbers, while squirrel, raccoon, and *Peromyscus* spp. had marked increases.
    - The raccoon population had the highest number captured in 2016 compared to previous years. Most likely, this is caused by the increase in juvenile individuals in the population

   - The *Peromyscus* spp. population may act in a cyclical manner but more data needs to be collected to confirm this
   - GSWA-CMA did not retain the population of subadult chipmunks marked in 2015 and had a new cohort of subadult chipmunks in 2016.
   - Age distribution of the chipmunk and squirrel populations over time indicates that chipmunk numbers decreased in 2016 because of loss in older individuals while squirrels have a younger, reproducing population.
2. My study documented a strong relationship between the chipmunk and squirrel populations. I accepted my alternative hypothesis that changes in the eastern gray squirrel and eastern chipmunk populations suggest a competitive relationship. Competition was indicated in both long and short term analyses.
   - Over time, the chipmunk and squirrel populations in my study displayed a strong negative correlation, when one population increases in numbers, the other declines.
   - In 2016, I determined that either squirrel or chipmunk activity dominates the trapping grids and they are not captured at equal rates
3. My study supported my alternative hypothesis of small mammal activity does differ among the sites at Drew University. Of note, squirrel activity was significantly higher at Hepburn Woods site than at the President's House Woods or the Zuck Arboretum sites.

4. My alternative hypothesis that there are differences in weight of small mammals based on location was supported as there are differences in weight of small mammals based on location. On average, Drew University chipmunks and squirrels weighed more than their respective captures at the GSWA sites.
5. I accepted my alternative hypothesis of being able to use weight as an indicator of age class in small mammals as I was able to use weight as an indicator of age class in chipmunks and squirrels. However, I was not able to use weight as an indicator of age class in *Peromyscus* spp.
    - A larger sample size is necessary to limit the impact of variation in individual weight and to determine if weight can be specific enough to aid in intermediate age class identification
6. Because there are no differences in weight of small mammals based on sex in adults, I did not reject my null hypothesis that there are no differences in weight of small mammals based on sex.
7. I could not reject my null hypothesis that the deer exclosure does not impact small mammal activity because I found that the deer exclosure does not impact small mammal activity over time.

Analysis of white-tailed deer (*Odocoileus virginianus*) population

The other portion of my thesis was the analysis of Drew University's white-tailed deer (*Odocoileus virginianus*) population over time. I monitored the white-tailed deer population in 2015 and 2016 using the methods in place for this long-term study, a fecal pellet index (FPI) and a visual census. I identified some interesting patterns in the white-tailed deer population using FPI.

Question 8: Deer fecal pellet index (FPI) and its implications for Drew University's population

Fecal pellet index (FPI) data provide evidence to estimate relative population abundance (Forsyth et al. 2007). Major population shifts in the white-tailed deer population at Drew University were implicated over time. From 2009-2011, the white-tailed deer population of Drew University experienced a decrease. The initial population decrease can be attributed to a combination of factors, including the installation of a deer fence around the Drew University Forest Preserve and exceeding carrying capacity, as 2009 had a record number of fecal pellets (Deffendall et al. 2011). The population decrease seen in 2012 is a consequence of an outbreak of epizootic hemorrhagic disease (EHD) that impacted Drew University's deer population, resulting in a 70% loss in population. The high percentage decline is not uncommon for white-tailed deer populations infected by EHD; early studies report mortality rates of over fifty per cent (Roughton 1975; Shope et al. 1960). However, total pellets counted in 2015 indicated that the white-tailed deer population made a complete recovery from EHD in four years, with a statistically significant difference in comparison to previous years' numbers. Despite this, the population peak experienced in 2015 is not significantly from that in 2009. The 2016 FPI indicated a significant decrease in pellets from values in 2015.

Both 2009 and 2015 peak years in population numbers were followed by great decreases in population. A potential explanation for these population drops is that the white-tailed deer population over exceeded their carrying capacity at Drew University's campus. It will be important to document how the deer population responds following this crash. Furthermore, it is of interest to see how long it takes for the deer population to recover given the time period of recovery from EHD.

Question 9: Visual Census

The visual census provides a visual documentation of the deer population. In 2015, the maximum deer seen per census was 7, while in 2016, the maximum deer seen on a given census was 4. Tilghman (1989) recommends that deer/ha stays below 0.067 to avoid detrimental impacts on forest composition and regeneration. The deer population in 2015 was 0.093 deer/ha, which is well above the suggested carrying capacity. In 2016, there was an estimated 0.053 deer/ha, which indicates that the deer population has dropped below Tilghman's (1989) maximum for sustainable levels. Nevertheless, these measurements of deer/ha were calculated using Drew University's full campus area, 75.28 ha, and this does not account for the Forest Preserve from which deer are excluded. Considering this, it does appear Drew's deer population still exceeds sustainable numbers but the decrease in 2016 is in agreement with the FPI.

The visual census allowed me to make conservative estimates of the age distribution of the white-tailed deer population on Drew's campus in 2015 and 2016. 2015 had a minimum of 2 adult doe, 1 adult buck and 2 fawns. In comparison, the 2016 population had a minimum of 1 adult doe and 3 fawns. Having observed 3 fawns but no bucks in 2016, this may indicate that our population lost some adult males. Correspondingly, McCullough (1999) found that because male deer are larger and require more resources, they have a higher mortality rate than those of the females.

Of note, there are some issues that should be acknowledged with this sampling technique, including confidence distance in deer visuals. In 2015, I determined that a conservative confidence distance in which I was accounting for all deer present was 25 m from the trail. This included 83% of the observations in 2015. However, in 2016, I only saw 44% of the deer recorded during the visual census within the 25 m confidence distance. This is in accordance with the FPI data that the deer population decreased in 2016, still, more than half of the deer that were documented were >25 meters away. This indicates that there may be deer on the campus that are not being counted in the census.

Most likely, the visual census under predicts the actual number of deer on Drew University's campus. In order to mitigate sampling error due to human eye, I suggest that binoculars be a requirement on future visual censuses. This could help samplers see further and account for patches that somehow the delineated trail is missing. Along with sampling error, there is a possibility that deer who reside on Drew's campus travel as it is not a closed system (Tierson et al. 1985). Tierson et al. (1985) suggests that white-tailed deer have a 225 ha summer home range, on average. Seeing as deer summer home range expands beyond the size Drew's campus, it is possible that the resident deer extend their home range beyond Drew on a given day and are then not recorded in a visual census. Although this dilemma may be mitigated by assuming the deer is accounted for on a different visual census, error could be further lessened by increasing the number of visual censuses performed as well as increasing the number of individuals counting deer. A study by Daniels (2006) used a ground census as one of the counting methods to estimate red deer (*Cervus elaphus*) population and had a high number of people counting deer in the census, for example, one census had 21 people. Daniels (2006) measured the coefficient of variation for this method to be 5-11%. There is clearly opportunity to improve on the methodology that I used in this census in the future and thus, I could be more confident in using the results from the visual census.

While acknowledging that there are flaws with the current visual census in terms of accurate representation of population numbers, there is value in these results. I kept the estimates conservative so that deer numbers were not overestimated. In light of this, the calculations of deer/ha and age distribution could only increase. With this in mind, it will be important to continue to monitor the deer population with the visual census in order to understand how the population changes over time, albeit conservatively.

Conclusions

8. I accepted my alternative hypothesis that there were changes in the white-tailed deer population over time because I found indication of changes in the population over time. Of note (according to FPI):
    - There was a marked decline in the population due to EHD in 2011. The population recovered in 3-4 years in 2015.
    - The population exhibited a crash in 2016, most likely due to an overshoot in carrying capacity
9. There were changes in the population and age distribution in 2015 and 2016 indicated by the visual census, thus, I accepted my alternative hypothesis that there were changes in the population and age distribution in 2015 and 2016 indicated by the visual census.
    - The visual census provided a rough measurement of the age and sex distribution of the deer population
    - The Drew University population exceeded carrying capacity in both 2015 and 2016 according to Tilghman (1989).
    - I established a confidence distance of 25 m.

The present findings of this thesis display the importance of long-term research with small mammal and white-tailed deer populations. Future research will continue to be essential in documenting how small mammal and white-tailed deer population dynamics change.

V. REFERENCES

Aguilar S. 2011. *Peromyscus leucopus* (online). Animal Diversity Web. [accessed 2017]. http://animaldiversity.org/accounts/Peromyscus_leucopus/.

Anderson RC. 1994. Height of white-flowered trillium (*Trillium grandiflorum*) as an index of deer browsing intensity. Biological Applications. 4(1):104-109.

Anderson RC, Strole TA. 1992. White-tailed deer browsing: Species preferences and implications forcentral Illinois forests. Natural Areas Journal. 12(3):139.

Armitage KB, Harris KS. 1982. Spatial patterning in sympatric populations of fox and gray squirrels. The American Midland Naturalist. 108(2):389-397.

Banks PB, Dickman CR. 2000. Effects of winter food supplementation on reproduction, body mass, and numbers of small mammals in montane australia. Canadian Journal of Zoology. 78(10):1775-1783.

Barr B, Wolverton S. 2014. The effects of population density on juvenile growth rate in white-tailed deer. Environmental Management. 54(4):897-907.

Bartmann RM, White GC, Carpenter LH. 1992. Compensatory mortality in a Colorado mule deer population. Wildlife Monographs. (121):3-39.

Beg MA, Hoffmann RS. 1977. Age determination and variation in the red-tailed chipmunk, *Eutamias ruficaudus*. Pacific Northwest Bird and Mammal Society. p. 26.

Bergman EJ, Doherty PF, White GC, Freddy DJ. 2015. Habitat and herbivore density: Response of mule deer to habitat management. Journal of Wildlife Management. 79(1):60-68.

Boerner RE, Brinkman JA. 1996. Ten years of tree seedling establishment and mortality in an ohio deciduous forest complex. Torrey Botanical Society. 123(4):309-317.

Broadbooks HE. 1958. Life history and ecology of the chipmunk, *eutamias amoenus*, in eastern washington. Museum of Zoology, University of Michigan. (103):5-56.

Brown BW, Batzli GO. 1984. Habitat selection by fox and gray squirrels: A multivariate analysis. The Journal of Wildlife Management. 48(2):616-621.

Brown JS. 1999. Vigilance, patch use and habitat selection: Foraging under predation risk. Evolutionary Ecology Research. 1(1):49-71.

Brown LG, Yeager LE. 1945. Fox squirrels and gray squirrels in illinois. Champaign : Illinois Natural History Survey.

Brunner JL, Duerr S, Keesing F, Killilea M, Vuong H, Ostfeld RS. 2013. An experimental test of competition among mice, chipmunks, and squirrels in deciduous forest fragments. PLOS ONE. 8(6):e66789.

Bunker A. 2001. *Peromyscus maniculatus* (online). Animal Diversity Web. [accessed 2017]. http://animaldiversity.org/accounts/Peromyscus_maniculatus/.

Byman D, Harding SD, Spear FW. 2013. Demographic effects of white-tailed deer (odocoileus virginianus) exclosures on white-footed mice (peromyscus leucopus). American Midland Naturalist. 170(1):171-183.

Collins BR, Anderson KH. 1994. Plant communities of new jersey: A study in landscape diversity. Rutgers University Press.

Côté SD, Rooney TP, Tremblay J-P, Dussault C, Waller DM. 2004. Ecological impacts of deer overabundance. Annual Reviews. p. 113.

da Fonseca GA, Robinson JG. 1990. Forest size and structure: Competitive and predatory effects on small mammal communities. Biological Conservation. 53:265-294.

Daniels MJ. 2006. Estimating red deer *Cervus elaphus* populations: An analysis of variation and cost-effectiveness of counting methods. Mammal Review. 36(3):235-247.

Deffendall R, Storie A, Weimer J, Belin C. 2011. Whitetail deer carrying capacity at Georgia Barrier Island. Georgia Journal of Science. 69(4):145-150.

DeStefano I. 2010. Abundance, richness, and activity of mammal populations in morris county, new jersey. Drew University.

Dice LR, Bradley RM. 1942. Growth in the deer-mouse, peromyscus maniculatus. Journal of Mammology. 23(4):416-427.

Drost CA, Fellers GM. 1991. Density cycles in an island population of deer mice, peromyscus maniculatus. Oikos. 60(3):351-364.

Dueser RD, Shugart HH. 1978. Microhabitats in a forest-floor small mammal fauna. Ecology. 59(1):89-98.

Flowerdew JR, Ellwood SA. 2001. Impacts of woodland deer on small mammal ecology. Forestry: An International Journal of Forest Research. 74(3):277-287.

Ford RG. 1981. Estimating small mammal ages from mark-recapture data. Journal of Zoology. 195(4):536-542.

Forsyth DM. 2005. Protocol for estimating changes in the relative abundance of deer in new zealand forests using the faecal pellet index (FPI). New Zealand Department of Conservation; [accessed].

Forsyth DM, Barker RJ, Grant M, Michael PS. 2007. Modeling the relationship between fecal pellet indices and deer density. Journal of Wildlife Management. 71(3):964-970.

Fox R, Dewey T. 2001. *Procyon lotor* (online). Animal Diversity Web.

Gannon WL, Sikes RS, The Animal Care and Use Committee of the American Society of M. 2007. Guidelines of the american society of mammalogists for the use of wild mammals in research. Journal of Mammalogy. 91(3):809.

Gaydos JK, Crum JM, Davidson WR, Cross SS, Owen SF, Stallknecht DE. 2004. Epizootiology of an epizootic hemorrhagic disease outbreak in west virginia. Journal Of Wildlife Diseases. 40(3):383-393.

Government of Alberta. 2005. Alberta wildlife animal care committee class protocol #007.http://aep.alberta.ca/fish-wildlife/wildlife-research collection/documents/WRClassProtocols007-SmallMammalHandlingTrapping.pdf

Great Swamp Watershed Association. 2017. Land conservation/cma. [accessed 2016]. http://greatswamp.org/what-we-do/land-conservation/.

Gustafson EJ, VanDruff LW. 1990. Behavior of black and gray morphs of sciurus carolinensis in an urban environment. The American Midland Naturalist. 123(1):186-192.

Hamilton WJ. 1936. The food and breeding habits of the raccoon. Ohio Journal of Science. 36(3):131-140.

Harestad AS, Bunnel FL. 1979. Home range and body weight--a reevaluation. Ecology. 60(2):389-402.

Ivan JS, Swihart RK. 2000. Selection of mast by granivorous rodents of the central hardwood forest region. Journal of Mammalogy. 81(2):549-562.

Jarnemo A, Liberg O. 2005. Red fox removal and roe deer fawn survival: A 14-year study. The Journal of Wildlife Management. 69(3):1090-1098.

Koprowski JL. 1994. Sciurus carolinensis. Mammalian Species. (480):1-9.

Korschgen LJ. 1981. Foods of fox and gray squirrels in missouri. The Journal of Wildlife Management. 45(1):260-266.

Kramer MT, Warren RJ, Ratnaswamy MJ, Bond BT. 1999. Determining sexual maturity of raccoons by external versus internal aging criteria. The Wildlife Society Bulletin (1973-2006). 27(1):231-234.

Lackey JA, Huckaby DG, Ormiston BG. 1985. Peromyscus leucopus. Mammalian Species. (247):1-10.

Loew SS. 2000. Sex-biased dispersal in eastern chipmunks, *tamias striatus*. Evolutionary Ecology. 13:557-577.

Lotze J-H. 1979. The raccoon (procyon lotor) on st. Catherines island, georgia. 4, comparisons of home ranges determined by livetrapping and radiotracking. New York, N.Y.: American Museum of Natural History.

Lotze J-H, Anderson S. 1979. Procyon lotor. Mammalian Species. (119):1-8.

Mares MA, Willig MR, Bitar NA. 1980. Home range size in eastern chipmunks, tamias striatus, as a function of number of captures: Statistical biases of inadequate sampling. Journal of Mammalogy. 61(4):661-669.

Martin K. 2015. Come and explore our new trail! The Great Swamp Watershed Association: Across the Watershed Blog. [accessed]. http://greatswamp.org/blog/come-and-explore-our-new-trail/.

McCullough DR. 1999. Density dependence and life-history strategies of ungulates. Journal of Mammalogy. 80(4):1130-1146.

McKeever S. 1958. Reproduction in the raccoon in the southeastern United States. Journal of Wildlife Management. 22(2):211.

McShea WJ. 2000. The influence of acorn crops on annual variation in rodent and bird populations. Ecology. 81(1):228-238.

McShea WJ, Rappole JH. 2000. Managing the abundance and diversity of breeding bird populations through manipulation of deer populations. Conservation Biology. 14(4):1161-1170.

Morris DW. 1979. Microhabitat utilization and species distribution of sympatric small mammals in southwestern Ontario. The American Midland Naturalist. 101(2):373-384.

Nupp TE, Swihart RK. 2001. Assessing competition between forest rodents in a fragmented landscape of midwestern usa. Mammalian Biology. 66(6):345-356.

Parsons AW, Simons TR, O'Connell AF, Stoskopf MK. 2013. Demographics, diet, movements, and survival of an isolated, unmanaged raccoon procyon lotor (procyonidae, carnivora) population on the outer banks of north carolina. Mammalia: International Journal of the Systematics, Biology & Ecology of Mammals. 77(1):21-30.

Pendergast TH, Hanlon SM, Long ZM, Royo AA, Carson WP. 2016. The legacy of deer overabundance: Long-term delays in herbaceous understory recovery. Canadian Journal of Forest Research. 46(3):362-369.

Pidduck ER, Falls JB. 1973. Reproduction and emergence of juveniles in *Tamias striatus* (rodentia: Sciuridae) at two localities in ontario, canada. Journal of Mammology. 54(3):693-707.

Pizzimenti JJ. 1979. The relative effectiveness of three types of traps for small mammals in some peruvian rodent communities. Acta Theriologica. 24(25):351-361.

Quist CF, Howerth EW, Stallknecht DE, Brown J, Pisell T, Nettles VF. 1997. Host defense responses associated with experimental hemorrhagic disease in white-tailed deer. Journal Of Wildlife Diseases. 33(3):584-599.

Reher S, Dausmann KH, Warnecke L, Turner JM. 2016. Food availability affects habitat use of eurasian red squirrels (*Sciurus vulgaris*) in a semi-urban environment. Journal of Mammalogy. 97(6):1543-1554.

Rooney T. 2001. Deer impacts on forest ecosystems: A north american perspective. Forestry. 74(3):201-208.

Roughton RD. 1975. An outbreak of a hemorrhagic disease in white-tailed deer in kentucky. Journal Of Wildlife Diseases. 11(2):177-186.

Rudd RL. 1955. Age, sex, and weight comparisons in three species of shrews. Journal of Mammalogy. (3):323.

Shelton AL, Inouye RS. 1995. Effect of browsing by deer on the growth and reproductive success of *Lactuca canadensis* (asteraceae). The American Midland Naturalist. 134(2):332-339.

Shope RE, McNamara LG, Mangold R. 1960. The virus-induced epizootic hemorrhagic disease of the virginia white-tailed deer (*Odocoileus virginianus*). Journal of Experimental Medicine. 111(2):155-170.

Sikes RS, Gannon WL. 2011. Guidelines of the American Society of Mammalogists for the use of wild mammals in research. Journal of Mammalogy. 92(1):235-253.

Snyder DP. 1982. *Tamias striatus*. Mammalian Species. (168):1-8.

Stephens RB, Anderson EM, Wendt SR, Meece JK. 2014. Field identification of sympatric *Peromyscus leucopus noveboracensis* and *P. maniculatus gracilis* in Wisconsin from external measurements. American Midland Naturalist. 171(1):139-146.

Stevens G, McCluskey B, King A, O'Hearn E, Mayr G. 2015. Review of the 2012 epizootic hemorrhagic disease outbreak in domestic ruminants in the united states. PLoS ONE. 10(8):1-11.

Stoeckeler J, Strothmann R, Krefting L. 1957. Effect of deer browsing on reproduction in the northern hardwood-hemlock type in northeastern wisconsin. The Journal of Wildlife Management. 21(1):75-80.

Svendsen GE, Wrazen JA. 1978. Feeding ecology of a population of eastern chipmunks (*Tamias striatus*) in southeast ohio. American Midland Naturalist. 100(1):190.

Tacutu R, Craig T, Budovsky A, Wuttke D, Lehmann G, Taranukha D, Costa J, Fraifeld VE, de Magalhaes JP. 2013. Human ageing genomic resources: Integrated databases and tools for the biology and genetics of ageing. Nucleic Acids Research. 41(D1):D1027-D1033.

Tierson WC, Mattfeld GF, Sage JRW, Behrend DF. 1985. Seasonal movements and home ranges of white-tailed deer in the Adirondacks. Journal of Wildlife Management. 49(3):760-769.

Tilghman NG. 1989. Impacts of white-tailed deer on forest regeneration in northwestern Pennsylvania. The Wildlife Society. p. 524.

Troyer EM, Cameron Devitt SE, Sunquist ME, Goswami VR, Oli MK. 2014. Survival, recruitment, and population growth rate of an important mesopredator: The northern raccoon. PLoS ONE. 9(6):e98535.

Tryon CA, Snyder DP. 1973. Biology of the eastern chipmunk, *Tamias striatus*: Life tables, age distributions, and trends in population numbers. Journal of Mammalogy. 54(1):145-168.

Uhlig HG. 1955. Weights of adult gray squirrels. Journal of Mammalogy. 36(2):293-296.

Waller DM, Alverson WS. 1997. The white-tailed deer: A keystone herbivore. Wildlife Society Bulletin (1973-2006). 25(2):217-226.

Wauters L, Dhondt AA. 1989. Body weight, longevity and reproductive success in red squirrels (*Sciurus vulgaris*). Journal of Animal Ecology. 58(2):637-651.

Webb SL, Kaunzinger CK. 1993. Biological invasion of the Drew University (New Jersey) forest preserve by Norway maple (*Acer-platanoides*). Bulletin of the Torrey Botanical Club. 120(3):343-349.

Wendland V. 1981. Cyclic population changes in three mouse species in the same woodland. Oecologia. 48(1):7-12.

Wolff JO, Dueser RD, Berry KS. 1985. Food habits of sympatric *Peromyscus leucopus* and *Peromyscus maniculatus*. Journal of Mammology. 66(4):795-798.

Yahner RH. 1978a. The adaptive nature of the social system and behavior in the eastern chipmunk, tamias striatus. Behavioral Ecology and Sociobiology. 3(4):397-427.

Yahner RH. 1978b. Weight gain of post-emergent juvenile *Tamias striatus*. Journal of Mammalogy. 59(1):196-197.

Yahner RH, Svendsen GE. 1978. Effects of climate on the circannual rhythm of the eastern chipmunk, *Tamias striatus*. Journal of Mammology. 59(1):109-117.

Yerger RW. 1955. Life history notes on the eastern chipmunk, *Tamias striatus lysteri* in central New York. The American Naturalist. 53(2):312-323.