The Effect of Oak Mortality and Other Environmental Variables on Small Mammal Communities

Sharon Newman

Submitted in partial fulfillment of the requirements of the degree of Masters of Arts in Conservation Biology under the Executive Committee of the Graduate School of Arts and Science

COLUMBIA UNIVERSITY IN THE CITY OF NEW YORK
October 15, 2010

Acknowledgements

I would like to thank those people who helped me through this thesis process. First of all, I thank my committee members, Dr. Katherine McFadden, Dr. Bill Schuster, and Dr. Dana Royer. These individuals have provided me with guidance and support, and without whom I would not have been able to complete this project. I must also express my heartfelt appreciation to those individuals who helped me to complete my field work: Heidi Smith, Amanda Geissler, and Tamathy Stage. Their hard work and dedication helped to bring this project from its initial stages to realization. I am very grateful to Stephanie Seto for her assistance in designing the experimental procedure for the current project.

I must also thank several individuals who helped me throughout the analysis portion of my thesis. Dr. Christine Johnson at the American Museum of Natural History and Dr. Michael Singer at Wesleyan University assisted in the identification of prey items, and I am grateful for their assistance. Dr. Matthew Palmer, Dr. Martin Lindquist, and Tyler McCormick at Columbia University also provided help with statistical analyses. I would also like to thank Katherine Pavlis at the Black Rock Forest Consortium and Sara Pace for their help in answering questions regarding the experimental study area. Finally, I thank Emily Schmidt, Kristin Winchell, Amy Kemp, as well as my family for all of their support, encouragement, and guidance.

Table of Contents

Abstract 1
Introduction 2
Methods 7
Study Area 7
Small Mammal Trapping 9
Vegetation Surveys 10
Arthropod Collection 11
Capture Analysis 11
Fecal Analyses 14
Environmental Variables 19
Results 20
Capture Analysis 20
Scat Analysis 23
Isotope Analysis 25
Environmental Variables 28
Discussion 30
Conclusion 43
Appendix A. 74
Appendix B 77
Appendix C 77

List of Figures

Figure 1. Map of the experimental plots 45
Figure 2. Total number of unique captures across the four treatment types. 46
Figure 3. Monthly Unique Captures 47
Figure 4. Stable isotopic ratios for predators and prey. 48

List of Tables

Table 1. Total number of individual species captured across the trapping period 49
Table 2. Number of captures by gender and age class across slope position and trapping month 50
Table 3. Number of unique single plot (SP) and unique multi-plot (MP) captures. 50
Table 4. Percent relative frequency of occurrence in fecal samples 51
Table 5. Tissue and blood stable carbon and nitrogen isotopic values 52
Table 6. Percent contribution of potential prey sources across treatment type 53
Table 7. Percent contribution of potential prey sources across slope position and over time 54
Table 8. Mean air and soil temperature (°C) over the trapping session 55
Table 9. Number of invertebrates collected across each treatment type 55

Abstract – Small mammal communities are often influenced by sudden or long-term environmental changes. However, despite the association between small mammals and their surrounding ecosystem, few studies have investigated these animals in a degraded temperate forest environment. The current study explores the effect of tree mortality due to invasive pathogens, pests, and diseases on small mammal abundances and foraging behavior. Trees were girdled at a study site in the Hudson Valley of New York to mimic the effects of such invasive agents on a deciduous forest environment. White footed mouse (*Peromyscus leucopus)* unique captures were found to increase in non-oak girdled treatment types, while Eastern chipmunk (*Tamias striatus*) captures decreased in oak girdled treatments. Additionally an analysis of small mammal diet revealed higher consumptions of invertebrates in girdled versus control plots, which is likely due to the increase in understory vegetative matter associated with the girdling event. By studying the effects of environmental change on small mammal communities, the current study attempts to demonstrate the complexity of northern temperate ecosystems, as well as the influence of tree mortality on small mammal populations.

Introduction

Widespread changes in forest structure and composition can result in sudden or long-term variations in small mammal population dynamics (see Pearce and Venier 2005, Larkin et al. 2008, Wang et al. 2009). Previous studies have shown a differential effect of habitat disturbance (e.g. forest fires, clear-cutting events, and invasive diseases) on various small mammal species. For example, species of mice (*Peromyscus* spp.) and voles (subfamily Arvicolinae) have been found to increase in abundance in response to forest fires (Sullivan, Lautenschlager, and Wagner 1999) and invasive pathogens (Meentemeyer et al. 2008); these environmental disturbances create an abundance of debris along the forest floor, which provides additional niche space for small mammal species (Sullivan, Lautenschlager, and Wagner 1999). However, evidence has shown that other species including chipmunks (*Tamias* sp.) and shrews (family muridae) have more variable reactions to environmental changes, ranging from positive to negative responses depending on the type and extent of such disturbances (Sullivan, Lautenschlager, and Wagner 1999).

Environmental variation may directly influence prey item abundance, thereby affecting small mammal population dynamics (e.g. Batzli 1977, McShea 2000, Clotfelter at al. 2007). Recent literature has focused on the influence of acorn production and oak tree (*Quercus* sp.) masting on small mammal abundances (McShea 2000). Several species of small mammals including mice, squirrels, and chipmunks rely on acorns as a source of food, especially during winter months (Batzli 1977, McShea 2000). Mast failure in oak-dominated forests is associated with increased mortality rates in white-footed mice populations during winter months, and subsequent population declines during the following spring and summer (McShea 2000). Abundance surveys on *P. leucopus*, *T. striatus,* and grey squirrel (*Scurius carolinensis)*

populations revealed an overall positive, yet highly complex, relationship between acorn production and small mammal population densities (McShea 2000). Declines in acorn production were found to have cascading effects; both small mammal communities as well as species from other trophic levels were either directly or indirectly influenced by acorn production (Clotfelter at al. 2007).

Changes in environmental conditions may not only affect small mammal abundances, but may also play a significant role in determining small mammal diet (Stephens and Krebs 1986, Drever & Harestad 1998). While predators are hypothesized to prefer food items which provide the maximum energy yield, theory predicts that generalist species will switch to alternative sources when preferred items are no longer available or when secondary sources become the most readily available food item (Stephens and Krebs 1986). Kelt et al. (2004) concluded that small mammal consumption of seeds is directly associated with variations in the availability of such prey items, caused by seasonal and microhabitat changes. Anthropogenic manipulations of environmental conditions have also led to alterations in food consumption; Sullivan (1979) found that introductions of food sources such as sunflower seeds led to a direct decline in the consumption of conifer seeds. Understanding foraging behavior can therefore provide scientists with a proxy for identifying changes in resource abundance as well as more widespread environmental change.

The current study examines the effect of environmental disturbances – specifically tree mortality caused by pathogens, pests, and other diseases – on both population dynamics and foraging ecology of small mammal communities. Several invasive plant diseases have been linked to widespread tree mortality throughout the United States (Meentemeyer et al. 2008). Chestnut Blight, Butternut Canker Disease, and Beech Bark Disease are known tree pathogens

that result in tree mortality. By subsequently decreasing the production of seeds and nuts, each of these diseases severely reduce food sources for small mammals (Davidson et al. 2005, Storer et al. 2005, Meentemeyer et al. 2008).

In recent years, Sudden oak death (SOD), caused by the water mold pathogen *Phytophthora ramorum,* has also resulted in rapid oak (genus *Quercus*) and tanoak (*Llithocarpus densiflorus*) tree mortalities throughout the western United States (Meentemeyer et al. 2004; Davidson et al., 2005; Fichtner, Lynch, and Rizzo 2007). Such widespread mortalities have been associated with frequent forest fires, as well as decreases in soil nutrient uptake and overall levels of biodiversity (Meentemeyer et al. 2004). Both native oak and tanoak trees, which are unable to resist infection by invasive pathogens, may be at high risk for mortality (McShea 2000, Clotfelter et al. 2007). Although sudden oak death is currently confined to western regions of the United States, ranging from Oregon to California, current models predict that the pathogen may begin to invade eastern forests during the next few decades (Meentemeyer et al. 2004, 2008). Oak trees are highly dominant in eastern forests, and SOD invasions have the potential to greatly affect these forest communities (Meentemeyer et al. 2008).

While many studies suggest that the majority of wildlife will be negatively affected by such pathogens (Davidson et al. 2005, Storer et al. 2005, Meentemeyer et al. 2008), further work will be necessary to compare small mammal populations both pre- and post-invasions. A few preliminary studies have attempted to predict the response of several species of small mammals to the introduction of Sudden Oak Death (Tempel et al. 2005, Apigian et al. 2005). Tempel et al. (2005) found that dusky-footed woodrat (*Neotoma fuscipes)*, brush mouse (*Peromyscus boylii*), and California pocket mouse (*Chaetodipus californicus*) populations in San Luis Obispo County, CA were higher at sites with higher levels of oak composition (high risk for SOD) compared to

those with fewer oaks (low risk for SOD, Tempel et al. 2005). In addition, others have found similar impacts of the plant pathogen, *Phytophthora cinnamomi*, on small mammal abundances, distribution, and diversity (Wilson et al. 1990) with strong negative correlations between small mammal population densities and the presence of the invasive agent (Wilson et al. 1990). However, past research has mostly focused on revealing general trends in "at risk" small mammal communities. In addition, few studies have examined the influence of invasive pathogens on both population abundance *and* feeding ecology (e.g. Apigian et al. 2005, Zwolak and Foresman 2007, Yarnell et al. 2007). Additional research will be necessary to identify species-specific responses of small mammal populations to invasive pathogens.

The objectives of this study were to quantify the variation in diet and abundances of small mammal communities in both normal and altered oak forest habitats. Unlike previous research, which has focused on single proxies as evidence of environmental change (e.g. Tempel et al. 2005, Apigian et al. 2005, Zwolak and Foresman 2007, Yarnell et al. 2007), the study combines multiple approaches (abundance data, fecal analyses, and stable isotope analyses). Fecal analyses were performed in order to identify prey items and served as independent complimentary data to the stable isotope analyses. However, scat analyses often over-estimate the contribution of certain prey items to small mammal diet, especially those items which are not readily digested (Nardoto et al. 2006). In addition, fecal analyses are also only effective at identifying short term feeding habits – on the order of days (Nardoto et al. 2006).

In order to identify how foraging might vary over a longer span of time, stable nitrogen ($\delta^{15}N$) and carbon ($\delta^{13}C$) isotopic analyses (SIA) were completed. SIA are a commonly used tool to discern feeding habits in wildlife because the isotopic ratios of consumers are related to those of their food sources (DeNiro & Epstein 1978, 1981). Differences in $\delta^{13}C$ ratios are often

indicative of the varying photosynthetic properties of plants; for example, C_3 plants are more depleted in ^{12}C when compared with C_4 plants (Farquhar, Ehleringer, and Hubick 1989, Nardoto et al. 2006). Nitrogen isotopic ratios can act a proxy for trophic level change, where a stepwise enrichment in $\delta^{15}N$ (or higher ratio of ^{15}N to ^{14}N) occurs at each trophic level (Hobson and Clark 1992, Hobson 1999, Bearhop et al. 2004).

Different tissues in the body have differing metabolic turnover rates, and therefore provide dietary information on varying timescales (Tiezsen et al. 1983). The metabolic turnover rate of skin is longer (on the order of months, Palerum 2005) when compared with the turnover rate of blood (on the order of weeks, Tiezsen et al. 1983). Studies of gerbil carbon fractionation rates revealed differences in metabolic turnover rates across 5 different tissue types, where hair>brain>muscle>fat>liver (half-life = 47.5, 28.2, 27,6, 15.6, 6.4 days, respectively; Tiezsen et al. 1983). Similar discrepancies in turnover rates were revealed for nitrogen fractionation, where liver turnover (~ 3 days) was faster than both muscle and blood turnover rates (2-3 weeks, respectively) for deer mice (*P. maniculatus*, Miller et al. 2008).

Several studies have successfully used fecal and stable isotopic analyses to reveal variations in feeding ecology as a result of environmental variation (e.g. McFadden 2006, Waddington 2008). Therefore, the goals of this experiment were to use such dietary proxies (fecal and stable isotopic evidence) to highlight changes in small mammal dietary patterns across both time and space.

Due to the highly inter-related nature of small mammal communities and their surrounding environment, it was hypothesized that generalist predator abundances will vary significantly between areas of high tree mortality and control plots. The following scenarios were proposed as possible outcomes for the experiment, based on evidence from the literature:

- Small mammal species will respond negatively to girdling effects (decreasing in abundance) as a result of reduced acorn and nut production in girdled plots (Batzli 1977, McShea 2000).
- Small mammal species will respond positively to the girdling effect (increasing in abundance) as a result of the increased niche space created by fallen logs and branches in girdled plots (Miller and Getz 1977, Kirkland 1990, Nordyke and Buskirk 1991)
- Small mammals will have differential responses to the girdling effect, based on species-specific differences (Sullivan, Lautenschlager, and Wagner 1999).

A second hypothesis was proposed, posturing that small mammals will alter foraging behavior in areas of environmental change (girdling effects, temporal changes, and other environmental variations) due to likely changes in prey availability.

Differential responses to oak (*Quercus* spp.) and non-oak tree mortality will reveal the influence that invasive pathogens may have on small mammals as well as provide insight into the effect that such diseases can have on forest ecosystems.

Methods

Study Area

Research was conducted in mixed deciduous forests of New York's Hudson Highlands at Black Rock Forest (41.41 N, 74.05 W). The forest consists of 15.5 km^2 of privately owned land located less than 1.6 km from the nearest town (Cornwall, NY; population 12,307 individuals, United States Census Bureau 2000). In June of 2008, a tree removal project was initiated, where selected trees were girdled to induce death. The girdling process was completed by chainsaw cutting around the circumference of the tree, and was done in an effort to mimic the effects of

invasive pathogens on a northeastern forest ecosystem. The 67,500 m^2 study site was divided into 12 plots, consisting of the following treatment types: (1) all non-oak trees girdled (N), (2) all oak trees girdled (O), (3) 50% of oak trees girdled (O50), and (4) no trees girdled (C). All treatment types were replicated three times throughout the experimental setting, once in each of three slope positions (lower, middle, and upper slope) (Figure 1). The size of the experimental plots (625m^2) is larger than the average home range of most small mammal species found at BRF (

Table 1).

Average elevation above sea level of lower slope plots was 152.4 m, compared to an elevation of 167.6 m for middle slope plots, and 182.9 m for upper slope plots. Plot elevation was estimated from the Black Rock Forest topographic map of the forest and the surrounding area. Each slope position includes all 4 treatment types (C, N, O, O50) in random organization (Figure 1). Lower slope plots are characterized by high soil moisture and organic matter levels. As one moves up the slope (to middle and upper plots), soils become increasingly drier with lower levels of organic matter (Black Rock Forest Consortium, unpublished qualitative data). Tree diversity is also highest at lower slope levels, whereas oak trees become the dominate tree species at middle and upper slope plots (Appendix A).

A quantitative assessment of vegetative ground cover across individual plots was completed by the Black Rock Forest researcher team in 2009. Their data revealed plots A1 and A3 vegetation to be dominated by Japanese Stiltgrass (*Microstegium vimineum*, Black Rock Forest Research Team, unpublished data). Plots A3, C1, and C4 had high concentrations of blueberry and huckleberry vegetation (*Vaccinium* and *Gaylussacia* spp., respectively). An overview of the experimental setting revealed B1, C2, and C3 to have a dearth in understory vegetation compared to the other plots (see Appendix A. Vegetative cover (%) across plots; vegetative cover was estimated for each plot in 2009 by summing together percent coverage for each species across plot quadrats (each plot was divided into ten 1 x 1 meter quadrats). Treatment type is presented beneath plot labels; all A, B, and C-plots represent A, B, and C-slope positions. Data was contributed to this thesis by the Black Rock Forest Consortium. for a complete vegetative analysis of the study area). In addition, a stream ran through plots A3, B3, C3 and C4.

Small Mammal Trapping

Trapping sessions were conducted from May to September 2009. A total of twenty collapsible Sherman live traps (7.5 x 7.5 x 25 cm) baited with rolled oats and peanut hearts were arranged in two concentric circles within each plot. The first concentric circle was 10 m from the center of the plot, and the second circle was 10 m away from the first circle. The distance between the outer rings of each concentric circle (i.e. between plots) was $35m^2$. Each trapping session was conducted simultaneously at a single slope position (e.g. A1 - A4) for 4 days/3 nights. All three slope positions were sampled once per month. In order to account for the various activity patterns of species caught, traps were checked twice daily: once, shortly after daybreak, and a second time, just before sunset.

Captured individuals were removed from Sherman traps and white-footed mice (*P. leucopus*) and eastern chipmunks (*T. striatus*) were marked with ear tags (model #1005-1, National Band and Tag Company, Newport, RI). Individuals were weighed to the nearest gram using a Pesola gram scale. Body length (mm), defined here as the distance from the tip of the nose to the start of the tail and tail length (mm), the distance from the base to the tip of the tail, was measured using a rigid ruler. Ear length (mm), the distance from base to the tip of the ear, and was also measured using a rigid ruler. However, due to the small size of *T. striatus* ears, and the relatively constant tail size, these measurements were not obtained for this species. Age and gender were also determined. For *P. leucopus*, age was determined by body coloration (grey = juvenile, mixed coloration = subadult, and brown = adult). *T. striatus* individuals less than 90g were classified as juvenile, while those above this weight were considered to be adults (Ford and Fahrig 2008). External genitalia were used to determine gender.

Blood and tissue samples were collected by taking a small skin sample (>5 mm) with shearing scissors after wiping skin with alcohol swabs. In an effort to minimize harm to captured individuals and maximize collection efficiency, skin tissues were obtained from the tails of *P. leucopus* captures and from the ears of *T. striatus* captures. Prior to incision, the tail (or ear) and scissors were cleaned with ethanol. All methods were approved by Columbia University's Institutional Animal Care and Use Committee (IACUC). A small sample of blood (approximately 4mm diameter blood drop) was then sampled directly onto a piece of glass fiber paper (Whatman, 2.4 cm in diameter). Upon return to the laboratory, tissue samples were then stored in the freezer (-20 ° Cs) and blood samples were placed in paper envelopes to sun dry for a period of 24 hours. Fecal samples were collected from the trap bag and the Sherman trap using forceps and refrigerated in cryovials. After processing, all animals were released at the point of capture.

Vegetation Surveys

Beginning in June, vegetative prey items were collected on the first day of each trapping session (totaling 15 days of collection). Such vegetation sampling was conducted by surveying the trapping area, and collecting all feasible mammalian prey items found within the 20 m trapping radius. A minimum of 5 prey items per prey species were collected over the entire study and stored in paper envelopes to avoid mildew growth. Prey items were then oven dried at 60 °C for 12 hours. Dried samples were used for two purposes: 1.) as voucher specimens to identify prey items in scat samples (to species or lowest level of identification) and 2.) for use in SIA.

Arthropod Collection

Leaf litter was collected twice during the experiment, once in August and again in September. During each collection session, a 1 x 1 m section was sampled at three pre-determined sites in each of the 12 experimental plots. Leaf litter was placed in plastic bags, closed, and mixed thoroughly and transferred to berlase funnels, suspended over containers filled with a 10 mL of 70% ethanol solution. Screens were attached to the top of the funnels to prevent prey items from escaping from the collection device. Incandescent light bulbs (60 watts) were positioned over the funnels in order to dry the leaf litter and to increase the number of prey items which fell into the containers of ethanol. After a three-day drying period, the screens were removed, and the ethanol containers were examined for the presence of invertebrates. Insects in the leaf litter which did not descend into the container, were manually extracted, and placed into the ethanol solution. All specimens were identified using the National Wildlife Federation Field Guide to Insects and Spiders of North America (Evans 2007) and in consultation with Dr. Michael Singer (Wesleyan University, Biology Department).

Capture Analysis

All statistical analyses were completed in R (Version 2.10.2009-12-10). P-values of 0.05 or less were considered to be statistically significant. Quantile-quantile probability plots were first generated for all data to assess normality. The total number of captures and the total number of unique individuals for each species were summed across plots and across all trapping sessions. Capture success was calculated based on the total number of captures per 100 trapping days (100 x number of captures/number of days). A one-way ANOVA was used to identify differences in capture success across treatment type, slope position, and over time (by month). It was not possible to use multi-factor models in the current study, due to the limited replication of variables; each treatment type (C, N, O, and O50) occurred only once at each slope position

(lower, middle, and upper slopes). Therefore, the interactive effects of such variables were unable to be successfully determined.

Unique captures were defined as those individuals that had not been previously captured during the current trapping year (May through September 2009). Only white-footed mouse (*Peromyscus leucopus*) and Eastern chipmunk (*Tamias striatus*) individuals were tagged, and therefore only these species were included in unique capture analyses. Both the short-tailed shrew (*B. brevicauda*) and masked shrew (*Sorex cinereus*) are venomous, and were not tagged during this trapping session. Other species were not tagged due to their infrequent capture. General linear models with Poisson distributions were then used to assess the effects of treatment type and slope position on *P. leucopus* and *T. striatus* unique captures.

Monthly unique capture data (or the number of captures unique to each month) was also calculated. A statistical analysis of the number of monthly unique captures was not possible due to the non-independent nature of this data. In addition, individuals were not recaptured with sufficient frequency over the 5 month trapping session to warrant the use of a repeated-measures ANOVA.

A Pearson's correlation was used to determine the association between body length and body mass for each species. Significant differences in unique *P. leucopus* body mass across treatment type, slope position, and over time (by month) were determined by using a one-way ANOVA. Differences in *T. striatus* body mass across treatment, slope position, and time were also examined using a Kruskal-Wallis analysis of variance by rank tests. A non-parametric Kruskal-Wallis test was used in place of a one-way ANOVA in cases where the data did not fit a normal distribution. Pearson's Chi square tests were used to determine if captures across

treatment type and slope position varied by gender, age (juvenile, sub-adult, and adult), and over time (trapping month) for both species.

Although the average home-ranges of unique species tested in the current study are smaller than the size of a single plot (50 m^2,

Table 1), many individuals were re-captured across multiple plots. Those individuals which were consistently captured in a single plot over the 5-month trapping period will hereafter be referred to as *single-plot* captures, while those individuals captured across multiple plots will be referred to as *multi-plot* captures. Therefore, in order to determine significant differences in single and multi-plot captures, statistical analyses were run comparing both populations. A Pearson's chi square test of independence was used to determine differences in the average frequency of unique captures for single and multi-plot captures across plots (for each species). Chi square tests were also used to determine *P. leucopus* and *T. striatus* differences in gender and age across plots. A student's t- test was run to determine differences in mass and body length between single and multi-plot unique captures for both species.

Small mammal species diversity estimates (which accounts for both abundance and evenness measures) were calculated for each of the 12 plots using the Shannon-Weiner Function (Krebs 1999):

$$H' = - \sum_i^s pi \ln (pi),$$

where H' is the Shannon-Weiner's diversity measure, s is the number of species i in the sample, and p_i is the proportion of species i in the total sample. A one-way ANOVA was generated to determine differences in species diversity across treatment type and slope position. Due to the fact that unique captures were identified for only 2 out of the 9 species captured (*P. leucopus* and *T. striatus)* at Black Rock Forest, total capture data for *all* species (rather than unique data) was used to determine diversity values. Therefore, while diversity estimates for the current study can provide insight into general trends across treatment and slope position, such evidence does not represent exact values.

Fecal Analyses

Fecal samples were processed in the laboratories at Columbia University and the Science Center at Black Rock Forest. Only unique samples from *P. leucopus* and *T. striatus* were analyzed. *S. cinereous* and *B. brevicauda* samples were not included in these analyses, due to the inability to determine unique specimens. Low capture rates and limited scat collection prevented analysis of the remaining captured species. Samples were first prepared for analysis by placing them a solution of 5 parts water and 1 part general detergent. The samples were shaken vigorously by hand, for one minute, and then left to soak in the detergent solution for a 12 hour period. The samples were individually washed through a 500 nm sieve (Hubbard, Berkeley, CA) in order to isolate prey items. Prey items were identified using a Stereo eighty microscope (SWIFT Instruments International) at 2x magnification, and were identified to the lowest taxonomic classification possible. Reference prey specimens were collected from the field to aid in the identification process. In addition, a subset of scat samples containing arthropod fragments were identified by Dr. Christine Johnson at the American Museum of Natural History in New York, NY; such identifications were used as reference source for arthropod identification in the remaining samples.

Invertebrates were generally identified to order and fruits (berries) were identified to genus. In cases where such identification was not possible, due to the highly digestible nature of certain prey items, the following general classifications were made: *other vegetation*, o*ther invertebrates*, and *other seeds*. These general prey classifications were mutually exclusive and did not overlap identified prey categories. Therefore, all prey items were identified with a *single* prey group classification.

The relative frequency of occurrence (%RFO=n/R*100) of each prey item in a scat was recorded by species, where *n* is the number of scats in which prey items are found, and R is the

total occurrence of food items (Kruuk and Parish 1981, Wu 1999). Due to the low frequency of certain invertebrate and berry prey items found in scat samples, prey items were pooled into ecologically meaningful guilds (berries, *other vegetation*, *other seeds*, and all invertebrates – the latter group was an aggregate sum of all identified invertebrate orders and *other* invertebrates). These guilds were consistent with the dietary history of small mammals found in the literature (Snyder 1982, Lackey et al 1985). Chi square tests (X^2) were used to determine if a significant difference existed in the frequency of occurrence of such guilds across treatment types, slope position, or over time.

The standardized niche breadth was calculated using Levins' measure (Krebs 1999):

$$B = 1/\sum p\, i^2,$$

where B is Levins' niche breadth measurement, p_i is the proportion of diet , and Y is the total number of samples. B can range from 1 to n, where n is the number of total resource states, and reaches a maximum value when each specimen use an equivalent number of resource states. Alternatively, B reaches a minimum value when each specimen uses a single resource state (Krebs, 1999). Here resource state is defined as prey category (Krebs 1999). Levins' niche breadth was then standardized from 0.0 to 1.0 using the following formula derived by Hurlbert (1978):

$$B_A = B\text{-}1/n\text{-}1,$$

where B_A is the standardized niche breadth, B is the original niche breadth, and n is the total number of resource states. Levins' measure was chosen because it focuses on dominant versus rare resources (Krebs 1999). A one-way ANOVA was used to determined differences in niche breadth across treatment type and slope position for *P. leucopus* and *T. striatus*.

Stable Isotope Analyses

All blood and prey item samples were prepared for stable isotope analysis by drying for 24 hours at 60°C. Due to the very small size of the tissue samples, lipid extractions was not completed. Prey items were ground using a mortar and pestle in order to homogenize for SIA. Samples were then weighed into tin capsules.

Stable isotopic analyses were conducted at the Boston University Stable Isotope Laboratory (BUSIL). Samples were combusted by a EuroVector Euro EA elemental analyzer at 1800 °C (BUSIL, Boston, Massachusetts, USA). Both N_2 and CO_2 gases were separated on a GC column and then passed through a GVI IsoPrime isotope ratio mass spectrometer. Water was separated from the samples through a magnesium perchlorate water trap. Ratios of $^{13}C/^{12}C$ and $^{15}N/^{14}N$ are expressed in delta notation (δ), in parts per mil (‰), using the following formulas for carbon and nitrogen isotopes, respectively:

$$\delta^{13}C=[(^{13}C/^{12}C)_{sample}/(^{13}C/^{12}C)_{standard}-1] \times 10^3$$

$$\delta^{15}N=[(^{15}N/^{14}N)_{sample}/(^{15}N/^{14}N)_{standard}-1] \times 10^3$$

All ratios are reported relative to secondary gas standards which have been previously calibrated to recognized international standards (National Bureau of Standards, Maryland, USA). These standards are inserted after every 11 samples in order to calibrate the system; standards for ^{13}C and ^{15}N are the Cretaceous marine fossil from the Pee Dee Belemnite formation, located in South Carolina and atmospheric nitrogen (VAIR), respectively. Sample precision for both ^{13}C and ^{15}N is 0.2 per mil (BUSIL, Boston, Massachusetts, USA).

A non-parametric Wilcoxon rank-sum test was performed on *P. leucopus/T.striatus* paired ^{13}C and ^{15}N values, in order to determine significant differences between blood and tissue types. Differences in ^{13}C and ^{15}N stable isotopic ratios across treatment types, slope positions, and over time were then determined by performing a one-way ANOVA; separate analyses were

performed for each species (*P. leucopus*, *T. striatus*) and tissue type (blood, tissue). Due to the low blood and tissue sample size, an analysis by trapping month (9 trapping days) was not ideal. Therefore, significant differences were measured across trapping season, which was divided into the following categories: *early field season* (the first 15 trapping days), *mid-field season* (days 16-30), and *late field season* (days 31-45), in order to increase sample size..

Stable isotopic analysis was also conducted on potential prey items collected from the field. Species of prey were grouped according to orders or family. A Grubb's test was used to search for outliers among the taxonomic prey categorization (i.e. order/family). Standard deviation was also calculated for each group in order to determine instances of high variation from the mean. Prey group stable isotope values from the literature were used to supplement the stable isotopic values for prey collected from the field. Using scat sample results as well as dietary information from the literature, prey groups were then clustered into ecologically meaningful guilds (e.g. invertebrates, berries, nuts, overall vegetation). A one-way ANOVA was used to determine if prey groups *within* each prey guild differed from one another. Prey groups which differed significantly were extracted from such guilds and made into their own guild. Differences in mean stable isotopic values *between* each guild were then tested by using a series of unpaired Student's *t*-tests.

Relationships between the isotopic signatures of animal tissue and diet are mediated by ^{13}C and ^{15}N depletion, caused by excretion or respiration. Therefore, the isotopic signature of animal tissue is often enriched compared with that signature found in the corresponding diet (DeNiro and Epstein 1978; 1981). Average fractionation values were obtained for the literature and added to prey item ^{13}C and ^{15}N values. Carbon fractionation values of +2‰ were used when fruit or nuts were consumed, and +1‰ for the consumption of invertebrates (Hilderbrand et al.

1996; Ben-David et al. 1997). Average nitrogen fractionation values of +2.03‰ were used for all prey types (Vanderlift and Ponsard 2003).

Multiple-source mixing models were then used to estimate the proportion of prey item contribution to small mammal diet (Kline 1990, 1993, Ben-David et al., 1997). The computer program Isosource version 1.13 (Phillips & Gregg 2001, 2003) uses an iterative approach to estimate diet contribution (Phillips and Gregg 2003). By comparing the isotopic signatures of the mixture (predators) with individual mean signatures of potential contributors (prey guilds), Isosource predicts the proportional contribution of each source, providing mean values as well as estimated contribution ranges (1-99th%, Phillips and Gregg 2003).

A dual-isotope (^{13}C, ^{15}N), 3-source (vegetation, invertebrates, nuts) model was selected for analysis, based on dietary evidence from the literature (e.g. Snyder 1982, Lackey et al. 1985) as well as fecal analysis results. An increment of 1% and a tolerance of 0.15 were initially set. However, in cases where no statistics could be generated (due to out of bound values), tolerance levels were incrementally increased by 0.05 to a maximum of 1.0.

Kruskal-Wallis rank sum tests were then used to determine statistical differences in ^{13}C and ^{15}N across plots for each treatment type, slope position, and trapping season category. Isotopic values from plots in the same treatment type/slope position/trapping season which did not statistically differ from one another were pooled to increase sample size. Mean values for each of these pooled isotopic ratios were input into Isosource to generate relative prey item contributions across treatment type, slope position, and over time.

Environmental Variables

Air temperature, soil temperature, and canopy openness data (the amount of light reaching the forest floor) was made available through the Black Rock Forest Consortium.

Although soil pH data was not available, pH values obtained from lysimeters were used as a proxy for soil acidity. Ground cover values (estimated by the Black Rock Forest research team, Appendix A) were determined by identifying percent understory ground cover for 20- 1x1 m^2 quadrats in each plot (A1-C4). For the current experiment, percent cover for each plot was then calculated by summing plot quadrat values, and dividing this number by 2,000% (or 20 x 100%). Significant differences across treatment type, slope position, and month were then identified for pH values (one-way ANOVA), soil temperature (Kruskal-Wallis rank sum test), and air temperature (Kruskal-Wallis rank sum test). Differences across treatment type and slope position were also determined for canopy openness (one-way ANOVA) and understory cover (Kruskal-Wallis rank sum test); an analysis of canopy openness and understory cover over time was not completed, due to the fact that monthly estimates for these variables were not available.

A Pearson's correlation test was then used to determine if significant associations occurred between *P. leucopus*/*T.striatus* unique capture frequencies and soil temperature, understory cover, and canopy openness. In order to determine such associations, unique captures for each species were first summed for each plot and compared against temperature, understory cover, and openness values for corresponding plots. These environmental variables were selected because they revealed the most significant differences across plots in the current experiment. Additionally, previous studies have shown soil temperature and vegetative coverage to be significantly associated with small mammal population dynamics (Ogilvie and Stinson 1966, Kritzman 1974, Tietje and Vreeland 1997, Tempel et al. 2008).

Results

Capture Analysis

Total Captures – A total of 1,593 small mammal captures representing 9 different species were made over 45 trapping nights (

Table 1). Capture success during each month ranged from 1466.7 (September) to 3173.3 (June) captures per 100 days (

Table 1). A one-way ANOVA revealed capture success to differ significantly across trapping month (ANOVA, $F = 23.85$, $df = 4$, $p < 0.001$), where June capture success was greater than July capture success; both June and July capture success was significantly greater than all other trapping months (Tukey post-hoc test, $p < 0.05$). A one-way ANOVA revealed no significant differences in capture success between either treatment types or slope position and capture success (ANOVA, $p > 0.05$).

During the 5 month trapping session, the majority of total captures (

Table 1) were Eastern chipmunks (71.2%, *T. striatus)* and white-footed mice (25.5%, *P. leucopus*). Other species captured included: Northern short-tailed shrews (37 captures, *Blarina brevicauda*), masked shrews (5 captures, *Sorex cinereus*), and woodland voles (6 captures, *Microtus pinetorum*). Single individual captures were found for the Southern-red back vole (*Clethrionomys gapperi*), long-tailed shrew (*Sorex dispar*), Northern flying squirrel (*Glaucomys sabrinus*), and long-tailed weasel (*Mustela frenata*).

Unique Captures across Treatment Type and Slope Position – Two-hundred and sixty three unique individuals were identified, which included 71 *P. leucopus* and 192 *T. striatus* individuals (as well as single captures of *C. gapperi*, *S. dispar,* and *G. sabrinus*). General linear models (GLMs) with poisson distributions revealed no association between *P. leucopus* or *T. striatus* unique captures and slope position (GLM, $p > 0.05$). However, significant associations across treatment type revealed higher numbers of *P. leucopus* captured in N treatment plots when compared with control plots (GLM, $Z = 2.04$, $p = 0.041$, Figure 2). *T. striatus* captures were significantly lower in O treatment plots compared with control plots (GLM, $Z = -2.18$, $p = 0.029$). No other significant effects of treatment type on unique captures were seen for either species (GLM, $p > 0.05$).

Captures Over Time – P. leucopus unique monthly captures (or a tabulation of those captures that were unique to each trapping month) did not fluctuate greatly over the 5 month trapping period, ranging from 17 to 25 individuals. However, *T. striatus* unique monthly captures ranged from 36 to 106 individuals; the highest number of unique monthly captures occurring in June, followed by July. The trapping months of May, August, and September had the lowest number of *T. striatus* unique monthly captures (Figure 3). Although unique captures were not identified for the other small mammal species, an analysis of total captures by month is presented in

Table 1.

Body Mass – Mean *P. leucopus* body mass was 20.02 g (range: 6-33 g). A positive association was found between *P. leucopus* body mass and both body length (Pearson's correlation, $T = 4.76$, $p < 0.001$). However, no significant differences were found in *P. leucopus* body mass across gender or age (ANOVA, $p<0.05$). A one-way ANOVA, revealed no significant effect of treatment type, slope position, or month on *P. leucopus* unique body mass (ANOVA, $p > 0.05$).

T. striatus body mass was found to be positively associated with body length (Pearson's correlation, $F = 5.01$, $p < 0.001$). Males were also found to have significantly higher body masses than females (ANOVA, $F=19.64$, $df=1$, $p=1.90xE^{-05}$). An analysis of *T. striatus* body mass by age was not attempted due to the fact that body mass was used to determine age class in the current experiment (Ford and Fahrig 2008). No significant differences in unique *T. striatus* body mass across treatment types or slope positions (Kruskal-Wallis, $p > 0.05$) were found. However, *T. striatus* body mass differed significantly over time (Kruskal-Wallis, $X^2 = 16.54$, $df = 4$, $p = 0.002$), with individuals captured in July having larger body mass than those captured in June and August (Nemenyi test, $p = 0.025$ and 0.030, respectively). All other changes in *T. striatus* body mass over the trapping session were found to be insignificant (Nemenyi test, $p > 0.05$).

Gender – P. leucopus had a male to female ratio of 8:5. No significant differences in gender across treatment type, slope position, or month were found for *P. leucopus* (X^2, $p > 0.05$, Table 2). The male to female ratio for *T. striatus* was 7:8 (Table 2). *T. striatus* showed no differences in gender across treatment type (X^2, $p > 0.05$). However, female *T. striatus* individuals were more frequently captured at lower slope plots, whereas male captures were more common at middle and upper slope plots ($X^2 = 10.9$, $df = 2$, $p = 0.004$). In addition, when compared to male

captures, more females were captured during the months of May and July, while fewer females were captured during the June trapping month (X^2 = 11.87, df = 4, p = 0.018).

Age – Adults made up the highest number of *P. leucopus* unique captures, followed by sub-adults and juveniles (Table 2); Pearson's chi square tests reveal no differences in *P. leucopus* age across treatment type (X^2, p > 0.05). However, higher numbers of juveniles were found in lower and upper slope plots, compared to lower numbers captured on middle slope plots; higher proportions of adults were found in lower and middle slope plots when compared with those captures at higher slope plots (X^2 = 16.08, df = 4, p = 0.003). Additionally, proportionally higher numbers of juveniles were captures in May, compared with proportionally higher numbers of sub-adults captured in July, and higher numbers of adults captured in August, compared with other age groups (X^2 = 16.12, df = 8, p = 0.041). All other differences across months and age categories were found to be insignificant (X^2, p > 0.05).

The ratio of T. striatus juveniles to adults was 11:9 (Table 2). *T. striatus* age did not differ across treatment type or slope position. The proportion of adult to juvenile captures was higher during May and July, and lower during the trapping month of June (X^2 = 10.96, df = 4, p = 0.027).

Single versus Multi-plot Unique Captures – A Pearson's chi square test of independence revealed no significant differences between single and multi-plot unique captures across plots for *P. leucopus* or *T. striatus* (X^2, p > 0.05, Table 3). A student's t-test also revealed no significant differences in mass or body length between single and multi-plot unique captures for either species (t-test, p > 0.05). Gender and age across plot did not vary significantly between single and multi-plot unique captures (X^2, p > 0.05).

Species Diversity – Shannon-Weiner diversity estimates (H') ranged from 0.40 to 0.95 per plot (Appendix B). A one-way ANOVA revealed no significant differences across treatment type or slope positions (ANOVA, $p > 0.05$). Although not statistically significant, diversity measures fluctuated across individual plots. Diversity values were highest in plots a1, a2, c3, and a4.

Scat Analysis

An analysis of the relative occurrence of prey items in scat samples (

Table 4) revealed *other vegetation* and *other invertebrates* to make up greatest contribution to *P. leucopus* diet (47.5% and 29.3%, respectively). Beetle fragments (Coleoptera,15.2%), blueberry seeds (*Vaccinium angustifolium* and/or *V. pallidum,* 3.0%) and *other seeds* (5.1%) were also found in *P. leucopus* fecal samples.

In contrast, a greater number of prey types were found in *T. striatus* fecal samples; while the majority of samples were composed of *other vegetation* and *other invertebrates* (39.0 and 28.9%), several berry species were present in the fecal samples including: blueberries (*Vaccinium angustifolium* and/or *V. pallidum*, 2.3%), huckleberries (*Gaylussacia baccata*, 1.4%), and wineberries (*Rubus phoenicolasius*, 1.1%). *Other seeds* were also present in *T. striatus* scat samples, and had a relative occurrence of 7.4%. Several orders and subfamilies of invertebrate species were also identified in *T. striatus* fecal samples which included: beetles (Coleopteran, 12.6%) ants (Hymenoptera, 4.0%), ticks (Ixodida, 1.4%), caddis flies (Trichoptera, 0.6%), and flies (Diptera, 1.1%).

Due to the low occurrence of several of the identified prey items, ecologically similar prey types were pooled for further analysis to create the following 4 prey guilds: berries, *other seeds*, *other vegetation*, and total invertebrates. These prey guilds were consistent with prey groups identified in the literature as making up a large portion of *P. leucopus* and *T. striatus* diet (Snyder 1982, Lackey et al. 1985). Relative frequencies of occurrences were then calculated for the four prey guilds. Differences in RFO were assessed using Chi square tests. *P. leucopus* diet differed significantly across treatment type (X^2= 27.73, df = 9, p = 0.001). Insect frequency of occurrence (RFO) was higher in girdled versus control treatments. The occurrence of *other seeds* followed a similar pattern, with low occurrences of *other seeds* in girdled treatments and no occurrences in control treatments. Berries were present only in those fecal samples collected

from N and C plots. Significant differences in *P. leucopus* diet across slope positions were also found (X^2 = 19.79, df = 6, p = 0.003); berries and *other seeds* were only found at lower and upper slope plots. Differences in RFO across month were also found to be significant, with berries and seeds present only in those scat samples collected in June and July (X^2 = 63.7, df = 12, p < 0.001).

No significant differences were found in *T. striatus* diet across treatment type (X^2, p > 0.05). However, *T. striatus* diet did vary significantly across slope position; animals captured on lower slope plots consumed the highest amount of berries and *other seeds*, when compared with the other slope positions (X^2 = 12.76, df = 6, p = 0.047). All other slope position differences were found to be insignificant (X^2, p > 0.05). Additional differences were found across month, with berries and *other seeds* found in scat samples collected during the months of June, July and August, and no seeds found during the other two trapping months. Insect RFO also decreased significantly in July and August when compared with the other months (X^2 = 94.12, df = 12, p < 0.001).

Niche breadth was calculated for each plot and ranged from 0.0 to 0.26 for *P. leucopus* and 0.15 to 0.40 for *T. striatus*. Niche breadth did not vary significantly across treatment types or slope positions for either species (ANOVA, p > 0.05).

Isotope Analysis

Blood versus tissue – A Mann-Whitney U test revealed that $\delta^{13}C$ isotopic values did not vary between blood and tissue for either species (Mann-Whitney U test, p > 0.05). However, significant differences were found in $\delta^{15}N$ across tissue types for both *P. leucopus* (Mann-

Whitney U Test, V = 91, $p < 0.001$) and *T. striatus* (Mann-Whitney U test, V = 168, $p < 0.001$, Table 5).

A one-way ANOVA revealed no significant differences in blood/tissue stable isotopes ($\delta^{13}C$ or $\delta^{15}N$) across treatment type, slope position, or over time for *P. leucopus* (ANOVA; $p > 0.05$). Analysis of *T. striatus* $\delta^{13}C$ blood and tissue ratios also revealed no significant differences across treatment type or slope position, for either blood or tissue samples (ANOVA; $p > 0.05$). However, *T. striatus* $\delta^{13}C$ blood values varied significantly over time (ANOVA, F = 5.597, df = 2, $p = 0.003$), with higher values in spring compared with summer months (Tukey post-hoc test, $p = 0.003$). *T. striatus* $\delta^{15}N$ blood values were also found to significantly differ over time (ANOVA, F = 7.83, df = 2, $p = 0.001$); higher values were found in summer when compared with spring (Tukey post-hoc test, $p < 0.05$).

***Prey Items* – A total of 59 prey items, representing 25 invertebrate and 34 vegetative specimens (Figure 4, Appendix C), were collected and processed for SIA. High variation in the $\delta^{15}N$ signature of diptera and lepidoptera (4.6 ± 3.12 and 3.6 ± 7.17 parts per mil, respectively), led to the substitution of values from the literature for data collected in the current experiment (Koenig et al. 2008, Bennet and Hobson 2009, Flaherty, Ben-David, and Smith 2010). Additionally, stable isotopic values found in the literature for Diplopoda (millipedes), Lumbricina (earthworms), and Coleoptera (beetles), orders which were not analyzed in this study, were added to the current set of isotopic values (Bennet and Hobson 2009). The following guilds were then created by aggregating ecologically similar prey types: invertebrates, depleted ^{15}N berries, enriched ^{15}N berries, and nuts (acorns and hickory). Fecal analyses revealed unidentified vegetative matter and seeds (both berry and *other*) to make up a significant portion of small mammal diet (**

Table 4). Therefore, an additional prey guild, vegetation (which included berries, and grass, and other seed producing plants), was incorporated in the current study, with stable isotopic values for vegetation taken from the literature (Darling and Bayne 2010). No significant differences were found *within* each prey guild (ANOVA, $p > 0.05$).

A series of unpaired Student's *t*-tests revealed significant differences in both ^{13}C and ^{15}N *between* the following guilds: invertebrates, depleted ^{15}N berries, enriched ^{15}N berries, and nuts (Student's t-test, $p < 0.05$). Vegetation values differed significantly from all guilds (Student's t-test, $p < 0.05$) except depleted ^{15}N berries (Student's t-test, $p > 0.05$).

Mixing model – Differing permutations of 3- and 4- source mixing models were generated which included a subset of the previously identified prey guilds. Only isotopically distinct prey sources were included in each mixing model. A final dual-isotope, 3-source model: vegetation, nuts, and invertebrates was selected based on dietary evidence from the literature (Snyder 1982, Lackey et al. 1985), the fecal analysis results (

Table **4**), as well as the overall mixing model results.

Regarding predator isotopic ratios, tissue and blood values which did not significantly differ from one another ($\delta^{13}C$) were pooled in order to increase sample size (with respect to species), while those values which did differ significantly across tissue types ($\delta^{15}N$) remained separated (Table 5). Kruskal-Wallis rank sum tests revealed no significant differences in predator $\delta^{13}C$ or $\delta^{15}N$ between plots for each individual treatment type, slope position, and over time (Kruskal-Wallis, $p > 0.05$) for either species. Therefore, predator values for each individual treatment type, slope position, and field season were also pooled before inputting mean aggregate $^{13}C/^{15}N$ values into Isosource (keeping *P. leucopus* and *T. striatus* values separate).

Mean estimated source contribution (%) to *P. leucopus* diet over the course of the entire study was 7.9 ± 1.6 vegetation, 51.1 ± 1.6 invertebrates, and 41.0 ± 1.5 nuts. Considering all treatment, slope position, and temporal variations, estimated ranges in diet for *P. leucopus* (Table 6, Table 7) are as follows: 0.0 to 20.0% (vegetation), 30.0 to 69.0% (invertebrates), and 31.0 to 53.0% (nuts). Invertebrates made up the highest mean percent contribution in N, O, and O50 treatments, low and mid-slope plots, and during the late trapping season when compared with corresponding treatment type, slope position, and time variables. Mean nut contribution was highest in C treatment types, mid- and upper slope plots, and decreased from spring to fall. Vegetation contribution was highest in C and N treatment types, lower slope plots, and during the early and mid-trapping seasons.

Overall, the mean estimated contribution (%) of source items to *T. striatus* diet was 15.3 ± 1.6 vegetation, 35.3 ± 1.6 invertebrates, 49.5 ± 1.4 nuts. Source contribution to *T. striatus* diet (Table 6, Table 7) ranged from 0.0 to 25.0% (vegetation), 21.0 to 51.5% (invertebrates), and 38.0 to 58.0% (nuts), taking variations in treatment type, slope position, and time into account. Mean

estimated invertebrate contribution was highest in O treatment plots, low and mid slope plots, and during the mid- and late trapping seasons; mean nut contribution did not fluctuate greatly across treatment type, but was highest in upper slope plots and the early trapping season. Mean vegetation contribution to diet was highest in C, N, and O50 treatment plots, upper slope plots, and the early and mid-trapping seasons (when compared against corresponding variables).

Environmental Variables

Variations in Environmental Variables over the Trapping Session – A one-way ANOVA revealed no significant differences in lysimeter pH values across treatment type, slope position, or by month (ANOVA, $p > 0.05$). However, soil temperature did differ across treatment type (Kruskal-Wallis, $X^2 = 262.3$, $df = 3$, $p < 0.001$, Table 8), where soil temperatures were significantly higher in C and N plots when compared with both O and O50 plots (Nemenyi test, $p < 0.05$). Soil temperature (Table 8) was also found to differ across slope positions (Kruskal-Wallis, $X^2 = 24.98$, $df = 2$, $p < 0.05$), with higher temperatures in lower slope plots, followed by mid- and then upper slope plots (Nemenyi test, $p < 0.05$). Soil temperatures increased from May until August, and then decreased in September (Nemenyi test, $p < 0.05$). No other differences across treatment type, slope position, or over time were found (Nemenyi test, $p > 0.05$).

Differences in air temperature (Table 8) occurred over time (ANOVA, $F = 152.0$, $df = 4$, $p < 0.001$), with significant increases in air temperature from May through August (Tukey post-hoc test, $p < 0.05$). All other monthly differences in temperature were found to be insignificant (Tukey post-hoc test, $p > 0.05$). No significant differences were found in air temperature across treatment type or slope position (ANOVA, $p > 0.05$).

Understory ground cover differed significantly across slope position (ANOVA, $F = 7.2$, $df = 2$, $p = 0.010$), with higher values in lower slope plots compared with both middle and upper

slope plots. (Tukey post-hoc test, $p < 0.05$). No significant differences in ground cover were found across treatment type (ANOVA, $p > 0.05$). Although the interactive effect of treatment type and slope position was unable to be determined, due to the lack of treatment repetition across each slope position, a general comparison of treatment type across slope position was made; N and O plots had the highest ground cover values when compared with corresponding O50 and C treatment types (with the exception of the middle slope plots, where C values were higher than O values; Appendix A). Canopy openness, or the amount of light reaching the forest floor, was found to differ significantly across treatment type (ANOVA, $F = 18.6$, $df = 3$, $p < 0.001$), with openness values higher in O plots versus all other treatment types (Tukey post-hoc test, $p < 0.05$).

Capture Frequencies and Environmental Variables – A Pearson's correlation revealed no association between either *P. leucopus* or *T. striatus* captures and soil temperature or understory ground cover (Pearson's correlation, $p > 0.05$). Similarly, no association was found between *P. leucopus* captures and canopy openness (Pearson's correlation, $p > 0.05$). However, a negative correlation was found between *T. striatus* captures and canopy openness, with higher numbers of captures at lower openness levels (Pearson's correlation, $T = -3.15$, $df = 10$, $p = 0.010$).

Discussion

Habitat Alteration and Small Mammal Captures – Small mammal populations utilize deciduous forests ecosystems for food, shelter, and breeding (Tietje and Vreeland 1997). The results of this study found that alterations in forest structure – determined in the current study by the girdling of 100% oak, 50% oak, or 100% non-oak trees – significantly influenced small mammal capture frequencies. Mean *P. leucopus* unique captures were found to be significantly higher in non-oak girdled sites (N) when compared with those captured in control plots (C).

Although results were not statistically significant, mean *P. leucopus* captures were also higher in oak-girdled (O) plots than control treatments. The significant increase in *P. leucopus* unique captures in N treatment types (and the increasing capture trend at O plots) may be associated with an overall increase in forest structural complexity (vertical structure) which often follows disturbance events (Miller and Getz 1977, Kirkland 1990, Nordyke and Buskirk 1991, Tietje and Vreeland 1997). Increased ground cover provides additional niche space for small mammal forest dwellers (Tietje and Vreeland 1997, Tempel et al. 2008). Small mammals use woody debris as a means of travel (Loeb 1993), as sites for nesting and foraging, and as places of refuge (Tietje and Vreeland 1997, Zollner and Crane 2003). Additionally, auditory predators such as owls are less likely to capture mammals which travel along fallen logs due to swifter and quieter prey movements (Zollner and Crane 2003). Therefore, it is possible that *P. leucopus* reacted positively to the girdling effect in N and O treatments.

Although no direct assessment of coarse woody debris (CWD) or vertical structure were available for the current study, previous research has revealed girdling efforts to produce an excess of snags (dead trees that have not yet fallen) and downed wood (which are produced by decaying trees), as well as promote increased understory growth (Noel 1970). Additionally, the small stem diameter of a subset of non-oak trees in the current study prevented widespread girdling efforts in N plots. In such instances, non-oak trees were cut down to initiate tree mortality. Cutting down these trees (in place of girdling) likely increased the amount of downed wood and overall understory vegetative debris in N plots, potentially resulting in the significant increase in *P. leucopus* unique captures at these treatment sites.

A comparison of vegetation across treatment type reveal further differences in understory cover between girdled and control treatments; the 100% girdled plots were found to have

generally higher mean understory coverage than corresponding C and O50 plots (although these findings were not statistically significant). In addition, canopy openness (the amount of light reaching the understory layer) can be used as a proxy for understory growth, where increased openness values are often correlated with increased CWD and understory growth (Carey and Johnson 1995, Mountford, Savill, Bebber 2006, Moores, Seymour, and Kenefic 2007). An analysis of canopy openness in the current study revealed increased values in O plots when compared with the other treatment types. Vegetative cover, like understory debris (e.g. CWD, downed wood), can provide additional niche space for small mammal forest dwellers (Tiejete and Vreeland 1997). Variations in understory cover across treatment type may have contributed to differences in *P. leucopus* unique captures.

Despite the suggested trend of increasing vertical structure in girdled plots, *T. striatus* unique captures were negatively affected by the girdling effect; significantly lower captures were found in O plots when compared with control treatments. Mean unique captures also increased incrementally from O50 to N to C treatments. Differential habitat usage among species may provide one explanation for why *T. striatus* captures did not increase in girdled plots, whereas *P. leucopus* populations seems to be positively influenced by the girdling effect (e.g. Dueser and Shugart 1978, Zollner and Crane 2003). Although *P. leucopus* capture frequencies are positively associated with woody debris and diversity/evenness of overstory and ground shrub cover, no significant effects of woody vegetation thickness, short woody stem diversity, or litter soil density were found on *T. striatus* abundance (Dueser and Shugart 1978); such evidence is partially supported in the current study, with no significant association found between *T. striatus* captures and understory ground cover. M'closkey and Fieldwick (1975) also demonstrated that *Tamias* species are most abundant in areas with high tree densities and are negatively associated

with an increased density of low-lying shrubs and other vegetation. Therefore, differential microhabitat utilization among *P. leucopus* and *T. striatus* communities could contribute to the capture differences found in the current study.

However, microclimate variations (where microclimate is defined as climatic conditions such as soil temperature, air temperature, and light, all measured at a local scale, Geiger 1965) may have also negatively affected *T. striatus* populations. Several studies have suggested that alterations in microclimate are directly responsible for a variety of ecosystem-wide patterns including: the distribution of organisms and overall canopy productivity (Shirley 1945, Chen et al. 1999). Oak-girdled plots in the current study had the highest soil temperature and amount of light reaching the forest floor (highest canopy openness) when compared with other girdled and control plots. Therefore, it is possible these microclimatic conditions may have altered the surrounding environment, thereby contributing to the low *T. striatus* capture rate in O plots. A few studies have revealed the independent effects of soil temperature and canopy openness on *T. striatus* populations (e.g. M'closkey and Fieldwick 1975, Landry-Cuerrier et al. 2008). Changes in soil temperature have been found to affect *T. striatus* torpor patterns; in times of resource scarcity, soil temperature can determine the maximum duration of *T. striatus* hibernations (Landry-Cuerrier et al. 2008). *T. striatus* above ground activity is also mediated by the amount of light reaching the forest floor (Kavanau 1969, Snyder 1982). Although such studies did not reveal specific negative effects of these environmental variables on *T. striatus* behavior, it is possible that changes in activity patterns could result in long-term consequences for *T. striatus*. In addition, without further research, it is impossible to predict the synergistic effects that these microclimate variables may have on such small mammal communities.

Significant decreases in *T. stratius* unique captures at O treatment types may also be associated with reduced acorn production in these plots. Although no quantitative assessment of acorn productivity is available, a gestalt overview of treatment types reveals a lower number of acorns produced in oak-girdled plots versus other treatments during the 2009 trapping season (Black Rock Forest Consortium, unpublished data). However, since both *T. striatus* and *P. leucopus* rely upon acorns as an important food source (Snyder 1982, Lackey et al 1985, Wrazen and Svedson, McShea 2000), it is unlikely that the most significant cause of *T. striatus* capture reductions was the decrease in acorn productivity. While the possibility exists that *T. striatus* are less effective at switching from acorns to additional food sources in times of acorn scarcity, the inability of the current study to assess acorn consumption in scat samples prevented such conclusions from being made.

Variations across Slope position – Several studies have revealed a direct correlation between changes in elevation and small mammal behavioral ecology (e.g. O'Farrell 1975, McCain 2005). However, significant findings in the current study are more likely to result from differences in microclimate and microhabitat (e.g. vegetative cover, vegetative diversity) variables across slope positions, rather than direct differences in elevation.

Variations in the proportion of *P. leucopus* age ratios may result from variations in soil structure, temperature, and moisture. Limited research has been completed on the effects of soil temperature on *P. leucopus* populations. However, a few preliminary studies reveal soil temperature to have a significant effect on *Peromyscus* (as well as other small mammal species) nest-building, breeding success, and habitat selection (Ogilvie and Stinson 1966, Kritzman 1974). Rhodes and Richmond (1983) determined the ideal soil temperature for nest-building in pine voles (*M. pinetorum*) to be 35°C or less. Similarly, Kritzman (1974) found that soil

temperature, along with soil moisture content and composition, were major factors in the selection of burrowing and nest sites for *Peromyscus* species. Differences in soil characteristics in the current study may have contributed to the *P. leucopus* age demographic patterns across slope positions. However, the high proportion of juvenile captures across both low soil temperature/high moisture (lower slope) and high temperature/low moisture (upper slope) plots, suggests that additional unidentified variables may play a significant role in *P. leucopus* breeding patterns.

Discrepancies in *T. striatus* gender ratios across slope position are possibly due to gender-biased dispersal patterns. Studies on *Tamias* behavioral ecology have revealed unequal ratios of male to female captures over time and space (e.g. Burt 1940, Yerger 1955). However, there is limited information on the agents responsible for differences in sex demographics. An investigation of gender-biased behavior in *Tamias* species reveals that natal males (or those born in a specific location) have greater dispersal patterns and are more likely to disperse than natal females (Loew 1999). Higher occurrences of female captures (and subsequently lower captures of males) at lower slope plots in the current study may suggest an increase in male emigration at those sites. Without a full analysis of male dispersal patterns, however, it will difficult to determine if natal males are emigrating from lower slope plots.

Species Diversity – Although species diversity did not differ significantly across treatment type or slope position, an analysis across individual plots reveals variations in mean diversity values. The highest species diversity estimates were found in the lower slope plots (specifically at plots A1, A2, and A4) as well as at plot C3. Increased understory coverage, high tree density, and high tree diversity at the lower slope plots may have contributed to the increased species diversity values (Appendix A, Hafner 1977, Dueser and Brown 1980). The abundance in low-

lying vegetation may have provided increased microhabitat niche space which is advantageous for small mammal populations (Hafner 1977, Dueser and Brown 1980). Increased diversity at plot C3 may have resulted from the presence of riverine environments. Previous research suggests that species abundance will increase near rivers and streams due to gaps in overstory coverage (or areas of increased sunlight and vegetative cover, Carey and Johnson 1995, Gehlhausen et al. 2000). Although overall data from plot C3 reveals no significant increase in canopy openness or ground cover when compared with the other plots, it is possible that an analysis of the area directly surrounding the stream (as opposed to an analysis of the entire plot area) may provide additional insight. A stream also runs through plots A3, B3, C4, which had correspondingly lower diversity measurements. Therefore, if species diversity is significantly associated with the presence of riverine habitats, it is likely that additional unidentified variables also play a role in determining species diversity.

Temporal Analysis of Unique Capture Data – T. striatus capture frequencies varied over the course of the field season. However, such variations may result from seasonal alternations in above-ground activity, rather than direct changes in population dynamics. While *P. leucopus* spends most of its active time on the forest floor (Lackey, Huckaby, and Ormiston 1985), *T. striatus* spends a significant portion of their active period in burrows (Panuska and Wade 1956). Therefore, increases in *T. striatus* below-ground activity could significantly lower overall capture frequencies. Several studies have revealed *T. striatus* individuals to have the highest above-ground activity during the late-spring breeding season and early to mid-autumnal foraging season (Yahner 1977, Lackey, Gregory, & Williams 1984). The unique monthly capture data in the current study reveals that *T. striatus* capture frequencies were highest during the month of June and September, and gradually decreased throughout the summer months. Such capture

results coincide with previously reported variations in *T. striatus* above-ground activity (Yahner 1977, Lacki, Gregory, & William 1984).

T. striatus activity patterns, however, do not always fluctuate on a population-wide level. Instead, activity patterns can be influenced by gender. For example, above-ground activity for male *T. striatus* individuals generally subsides during summer months (Schooley 1934, Lacki, Gregory, & William 1984, Dunford 1972, Yahner 1977). Explanations for this lull have included low energy demand for male *T. striatus* individuals (Yahner 1977), adverse or rapidly changing climatic conditions during mid-summer (Schooley 1934), and an overall decline in food resources (Yahner 1977). Alternatively, females increase their above-ground activities during the late summer months (Schooley 1934, Yerger 1955, Lacki, Gregory, William 1984), in order to partake in post-breeding activities (Yerger 1955). In the current study, there was a significantly higher number of female captures during the trapping month of June when compared with male captures. Such discrepancies in the ratio of male to female captures during this trapping month may reflect the differential activity patterns of male and female *T. striatus* individuals that are often found during mid to late-summer months (Schooley 1934, Yerger 1955, Lacki, Gregory, William 1984).

Furthermore, the current study revealed fluctuations in age level capture frequencies for both *T. striatus* and *P. leucopus*. Such variations in age demographics over time may reflect these species breeding patterns. Both the *T. striatus* and *P. leucopus* breeding cycle occurs biannually, the first mating period occurring during spring (March-April), and the second occurring during mid-summer (July-August; Schooley 1934, Yerger 1955, Burt 1940, Snyder 1982, Lackey, Huckaby, Ormiston 1985). Juvenile *P. leucopus* leave the nest 2 to 4 weeks after birth, whereas young *T. striatus* emerge from the ground around 4 to 7 weeks (Yerger 1955,

Yahner 1977, Huckaby, Ormiston 1985). Therefore, an increase in juvenile captures for both species is expected to occur during early/mid-summer and again in early fall (Yerger 1955, Yahner 1977, Huckaby, Ormiston 1985). The current study reveals an increase in the ratio of *P. leucopus* unique juvenile captures during the month of June, and a corresponding increase in *T. striatus* juvenile captures during the months of May and June. The increased ratio of juvenile to adult captures during the mid-summer months may be indicative of an early-spring breeding season for both species. Although the study was terminated in September, an extension of the trapping season may have revealed a second spike in juvenile captures during early-autumn.

Increases in *T. striatus* body mass found for those individuals captured during the month of July may also be influenced by breeding activities during mid-summer months (Snyder 1982, Lackey et al. 1985), where female sciurids were found to gain body mass during their gestation period (Humphries and Boutin 1999). Additionally Brenner and Lyle (1975) found *T. striatus* body mass to be associated with photoperiodic conditions, where *T. striatus* mass was found to increase during the mid summer months (under natural light conditions). Such results are consistent with the increase in mean body mass found in the current experiment.

Dietary Analyses – Higher proportions of invertebrates were found in *P. leucopus* scat samples collected in girdled compared to control plots. Prey item contribution to small mammal diets generated from isotopic ratios supported such findings, where estimated invertebrate contribution to diet was highest in N, O, and O50 plots for *P. leucopus* and O plots for *T. striatus*, when compared with control plots for both species. Generalist predators are likely to consume the most readily available prey, switching to alternative sources of food when preferred items are no longer available (Hansson 2004, Boonstra and Krebs 2006). The increase in invertebrate consumption in girdled plots may therefore be associated with an overall increase in invertebrate

abundances following girdling events. Increases in understory ground cover are often associated with correspondingly higher arthropod densities (Jabin et al. 2004, Horn and Hanula 2008); high levels of coarse woody debris and ground vegetation create additional niche space for invertebrate species as well as improve soil conditions for arthropod communities (Savely 1939, Loeb 1993, Stevens 1997, Jabin et al. 2004). Therefore, girdled plots, which most likely contain high levels of ground cover, are expected to have correspondingly higher levels of invertebrates. An analysis of invertebrate captures across treatment types supports such theory, with O50 and N plots having the highest number of invertebrate captures (Table 9).

Variation in berry consumption also fluctuated across treatment type and slope position. Fecal results reveal the highest consumption of berries to occur at C and N treatment types and at upper slope plots for both *P. leucopus* and *T. striatus*. However, it is likely that ecological factors rather than direct experimental manipulation may have resulted in such findings. Fecal evidence reveals a correlation between the presence of berries in small mammal diet and the occurrence of berry-producing vegetation at specific plots. For example, all occurrences of Ericaceae (blueberry and huckleberry) seeds were found in scats that were collected from plots A3, A4, B4, and C4, which contain populations of *V. angustifolium* (Lowbush blueberry), *V. pallidum* (Blue Ridge Blueberry), and/or *G. baccata* (Black Huckleberry). The presence of such vegetation may be partially due to the close proximity of water to the aforementioned plots; a stream runs through plots A3 and B3, as well as cuts through the corners of C4. The presence of streams can create gaps in the overstory layer which are likely to promote understory plant growth (Carey and Johnson 1995). In addition, the synergistic effects of soil temperature and pH values across plots may have had an effect on understory growth (Brosofske, Chen, and Crow

2001). However, the specific effects of these variables on plant growth are beyond the scope of the current experiment.

Changes in diet over time – T. striatus $\delta^{13}C$ isotopic blood values fluctuated over the field season, where samples collected during the early field season were enriched compared to those collected the mid-field season. Such variation in $\delta^{13}C$ may be reflective of seasonal changes in the surrounding environment and concomitant shifts in prey availability. While forest environments with low light levels (i.e. high canopy coverage) contain plants which are enriched in ^{13}C, high-light vegetation (found in areas of low canopy coverage) are often depleted in the heavier isotope (Hogberg et al. 1995, France 1996, Evans and Belnap 1999, Darling and Bayne 2010). Such differences in isotopic values are the result of increased leaf litter decomposition in low light environments. The carbon dioxide (CO_2) released from soil respiration is highly depleted in ^{13}C. Therefore, plants which inhabit low-light environments are also likely to have lower $\delta^{13}C$ ratios (Hogberg et al. 1995, France 1996). Although no data on canopy coverage over time was made available for the current study, an assessment of northern temperate forest ecosystems reveals an increase in overstory cover for deciduous plants during the early summer months (Blackburn and Milton 2010). The increase in canopy coverage from spring to summer could help to explain the variations in *T. striatus* $\delta^{13}C$ isotopic enrichment over time which were found in the current study.

Temporal differences in blood $\delta^{15}N$ isotopic values were also found for *T. striatus* and may indicate variations in consumer trophic level patterns. Nitrogen isotopic values will become increasingly enriched as the consumer feeds on prey items that are higher up on the food chain (DeNiro and Epstein 1980, Minagawa and Wada 1984). An analysis of *T. striatus* blood samples revealed depleted ^{15}N isotopic values in those samples collected during the early field season

compared with those collected during the mid-field season. Due to the fact that blood stable isotopic values provide dietary information on the order of many weeks (Tieszen et al. 1983), such results may indicate a significant change in consumer trophic level patterns between late spring and early to mid-summer.

Moreover, differences in turnover rates of tissue and blood may explain isotopic variations in $\delta^{15}N$ blood and tissue values found in the current study. Due to the fact that tissue and blood provide dietary information on varying timescales (tissue = months, blood = weeks, Palerum 2005, Tiezsen et al. 1983), it is likely that isotopic discrepancies between tissue types may reflect a significant shift in consumer trophic level patterns over time. The current data reveal tissue $\delta^{15}N$ to be significantly higher than blood values, which suggests that there may be a temporal variation in prey item consumption.

The mixing model results also reveal temporal fluctuations in dietary patterns, with the highest vegetative contribution to small mammal diet occurring during the mid-trapping season for *P. leucopus* and the early and mid-trapping season for *T. striatus*. An analysis of the fecal results by trapping season provides further insight into such findings; vegetation (berries, seeds, and unidentified vegetative matter) was found most frequently in those fecal samples collected during June and July (the mid-trapping season months). It is likely that temporal fluctuations in vegetative consumption are mediated by the abundances and availability of vegetative prey items in the environment (Mackay and Kalff 1969, Johnson 1975). Fruiting plants (e.g. *Vaccinium* and *Gaylussacia*) begin to produce ripened fruits and berries during the mid-summer months (Oliver 1972, Elias and Dykeman 2009), the time at which small mammals in the current study were found to be consuming the highest proportion of berries and *other seeds* (

Table 4).

The mixing model results revealed the highest estimated nut (hickory and acorn) contribution to small mammal diet to occur during the early trapping season, with lower estimates during the mid- and late- trapping seasons. Nuts were unable to be identified in the fecal samples, therefore only general trends can be gathered from these findings. While no extensive survey of nut production was available for the current trapping location, the literature suggests that nut production peaks during the during fall season (beginning in late September and October, Oliver 1972, Elias and Dykeman 2009).). Therefore, nuts are expected to make up a small proportion of small mammal summer diet (Oliver 1972, Elias and Dykeman 2009), as is represented by the mixing model evidence.

Similarly, fluctuations in small mammal invertebrate consumption are likely associated with prey item availability. The mixing model results reveal higher levels of estimated invertebrate contribution to diet during late trapping season for both species when compared with early-trapping season data. The fecal results partially support such data, with higher invertebrate frequencies during August and September (the late trapping season) when compared with the mid-trapping season. Higher invertebrate abundance and species diversity have been reported during the late summer months when compared with early summer in similar forest environments (Mackay and Kalff 1969, Johnson 1975). Temporal fluctuations in invertebrate populations are most likely due to changes in air temperature and rainfall patterns throughout the summer (Matthews and Matthews 1970). However, a further investigation of the relationship between temperature/rainfall and invertebrate populations was again beyond the scope of the current experiment.

Caveats – While girdling is an effective means of causing tree mortality, the negative effects of girdling are not immediate. A survey of the experimental plots (completed in early spring 2009) revealed 7.8% of girdled trees to have survived in O plots, 14.6% in O50 plots, and 24.3% in N plots. Trees may survive post-girdling anywhere from one to four years (Baldwin 1934), depending on their overall girth, size, and species (Noel 1970). The full effects of girdling, such as increases in fallen branches, downed wood, and leaf defoliation may not have occurred at the onset of the current trapping experiment, which took place only one year after the girdling event occurred. Future analyses of small mammal populations at Black Rock Forest will help to determine the long-term effects of girdling events.

The number and quality of treatment replicates may have also affected the statistical significance of variation between treatment types. The experimental plot set-up contains three replicates of each of the four treatment types (C, N, O, and O50). However, each plot varies in both elevation and/or microhabitat complexity (e.g. presence of riverine environments and vegetation). While the current study attempted to isolate such variables – by analyzing the influence of slope position and other environmental variables on small mammal communities – the effect of inter and intra-plot variations (as opposed to treatment-level variations) could not be completely eliminated. Therefore, limited significant findings do not necessarily rule out the possibly for differences between treatment types. Further analysis with an increased number treatment replicates may reveal supplemental findings.

Small mammals may have also migrated across plots (Table 3), affecting the independent nature of each treatment type. Although the current study compares single and multi-plot unique captures, the study was unable to identify *all* individuals which traversed multiple plots.

Therefore, a subset of the single-plot captures may have ventured into two or more plots during the trapping session.

Deer exclosures (10 m^2 plastic mesh wire fences) were located near the northern perimeter of each of the plots. Although trapping efforts were not completed inside the exclosures, small mammals were able to pass between the exclosures and the central plots. Additional research investigating differences in small mammal activities inside deer exclosures and in the surrounding plot area will need to be completed in order to successfully investigate the effect of deer exclosures on the small mammal populations.

Conclusion

In recent years, invasive pests and pathogens have devastated temperate forest ecosystems, causing widespread tree mortality (Meentemeyer et al. 2004; Davidson et al., 2005; Fichtner, Lynch, and Rizzo 2007). Small mammals are excellent indicators of ecosystem change due to their rapid responses to alterations in the surrounding environment (Monthey & Soutiere 1985, Zwolak and Foresman 2007, Yarnell et al. 2007). Although preliminary data has suggested an overall negative effect of invasive diseases on small mammal populations (Davidson et al. 2005, Storer et al. 2005), the results of the current study, in combination with recent research on Sudden Oak Death (Tempel 2004), suggest that the effects may be more variable and complex than previously thought. Responses to invasive agents will most likely differ according to species, with *Tamias* populations decreasing in areas of high oak tree disturbance. *Peromyscus* communities are likely to benefit from high tree mortality, which may lead to concomitant increases in black legged ticks (which are vectors for Lyme and other diseases, Donahue, Piesman, and Spielman 1987, Schauber, Ostfeld, and Evans 1995). Therefore, highly degraded sites could potentially pose a threat to human health. Small mammal dietary responses are likely

to vary according to the distribution and availability of food items. Such findings indicate that small mammal species may be able to adapt to rapidly changing food sources caused by alterations in the surrounding environment. However, additional research will need to be completed in order to determine the long-term consequences of environmental degradation at Black Rock Forest. Only through continued investigation of the current study site, as well as similar degraded environments, will scientists be able to better understand the effects of environmental change on forest ecosystems and small mammal populations.

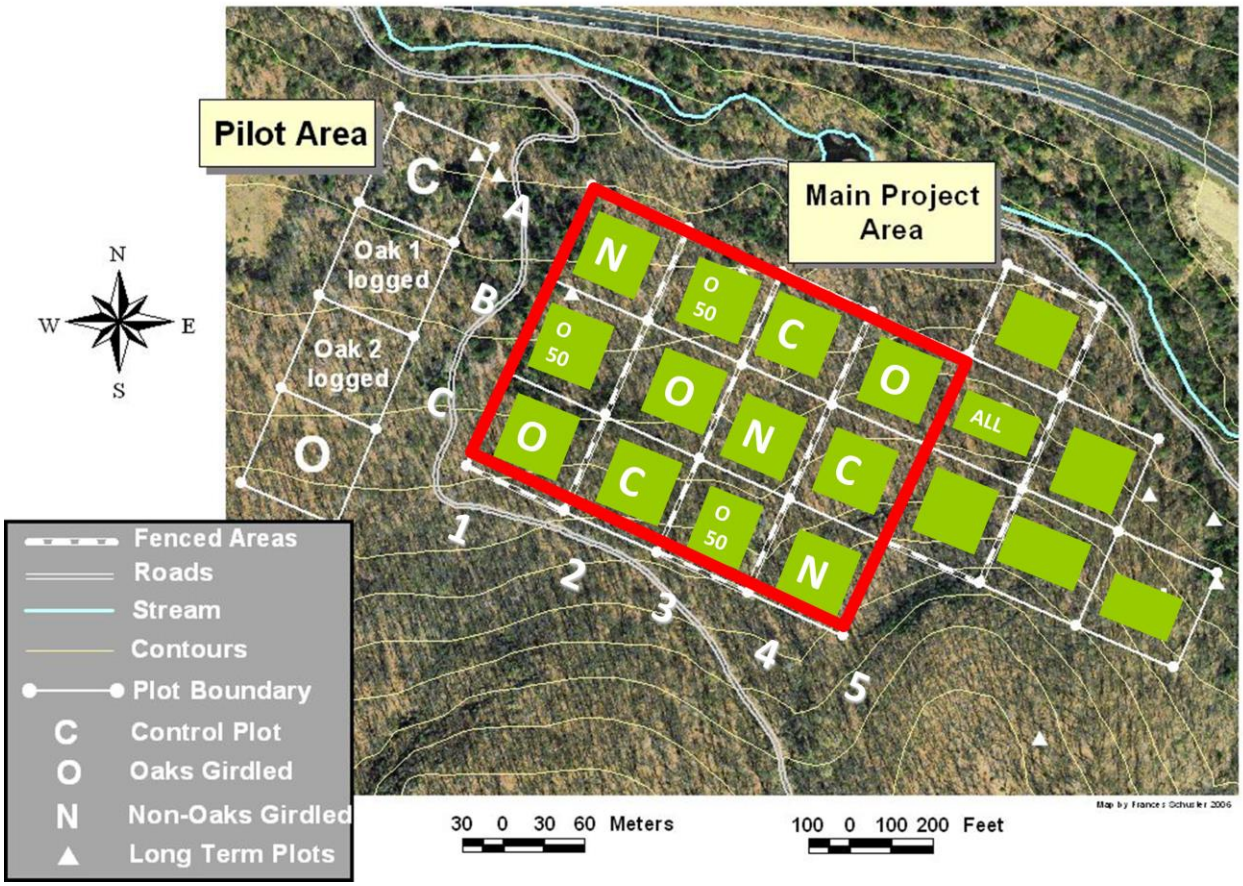


Figure 1. Map of the experimental plots contributed to the current thesis by the Black Rock Forest Consortium. Trapping experiments were conducted within the area marked by a red border. Upper, middle, and lower slope plots are represented by A, B, and C plots on the map, respectively. External plots represent varying levels of degradation (e.g. All plots are those where all trees were girdled).

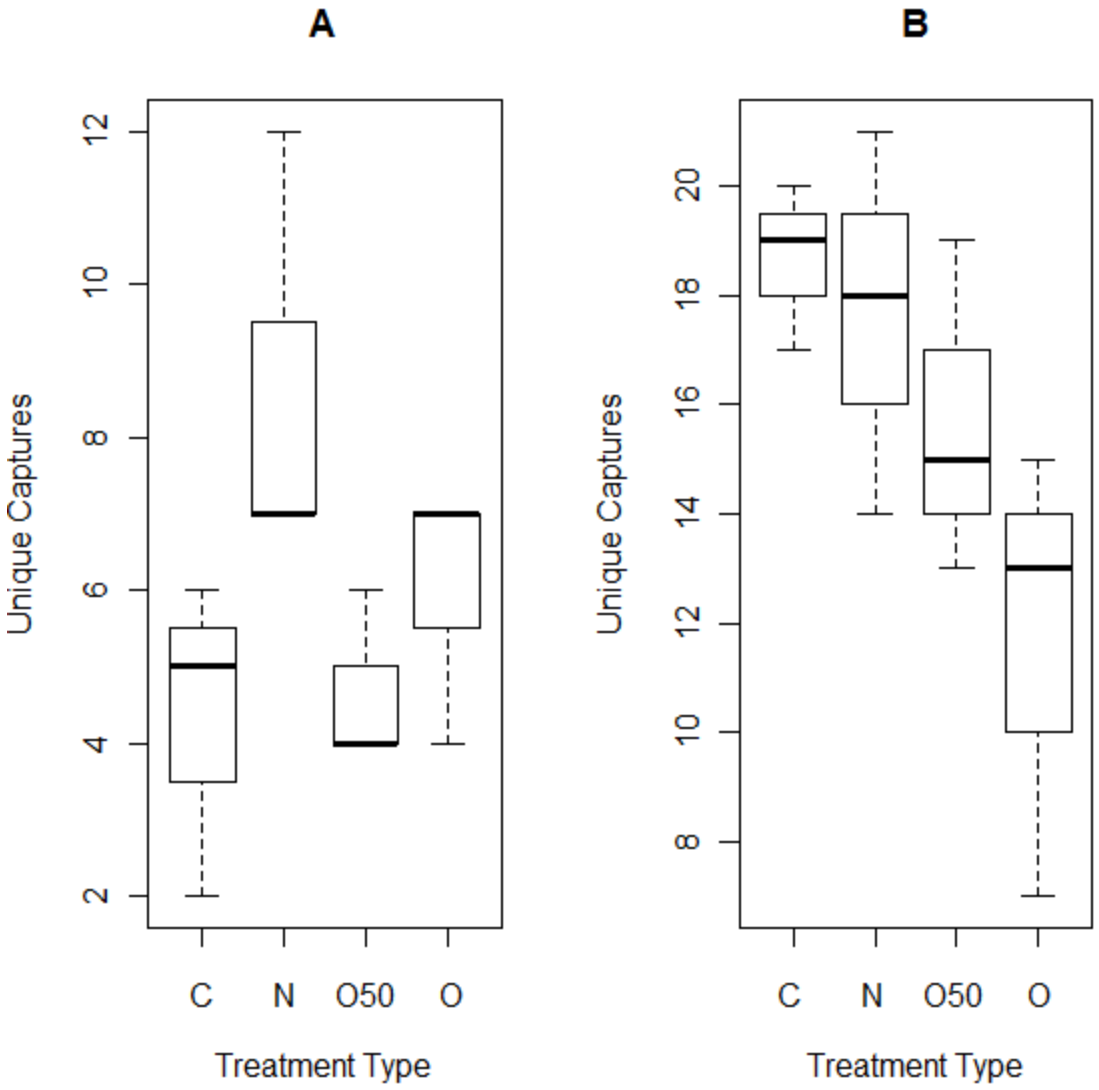


Figure 2. Total number of unique captures for A) *P. leucopus* and B) *T. striatus* across the four treatment types (C = control plots, N = non-oak girdled plots, O50 = half oak-girdled plots, and O = oak-girdled plots) at Black Rock Forest, NY from May to September (2009). Treatments are displayed from least disturbance to most disturbance. The whiskers represent minimum and maximum capture values for each species. Standard deviation for *P. leucopus* is as follows: C = ±1.08, N = ±3.51, O = ±1.73, and ±O50 = 1.15 captures. *T. striatus* standard deviation for each treatment type is: C = ±1.53, N = ±3.51, O = ±4.16, and O50 = ±3.06 captures).

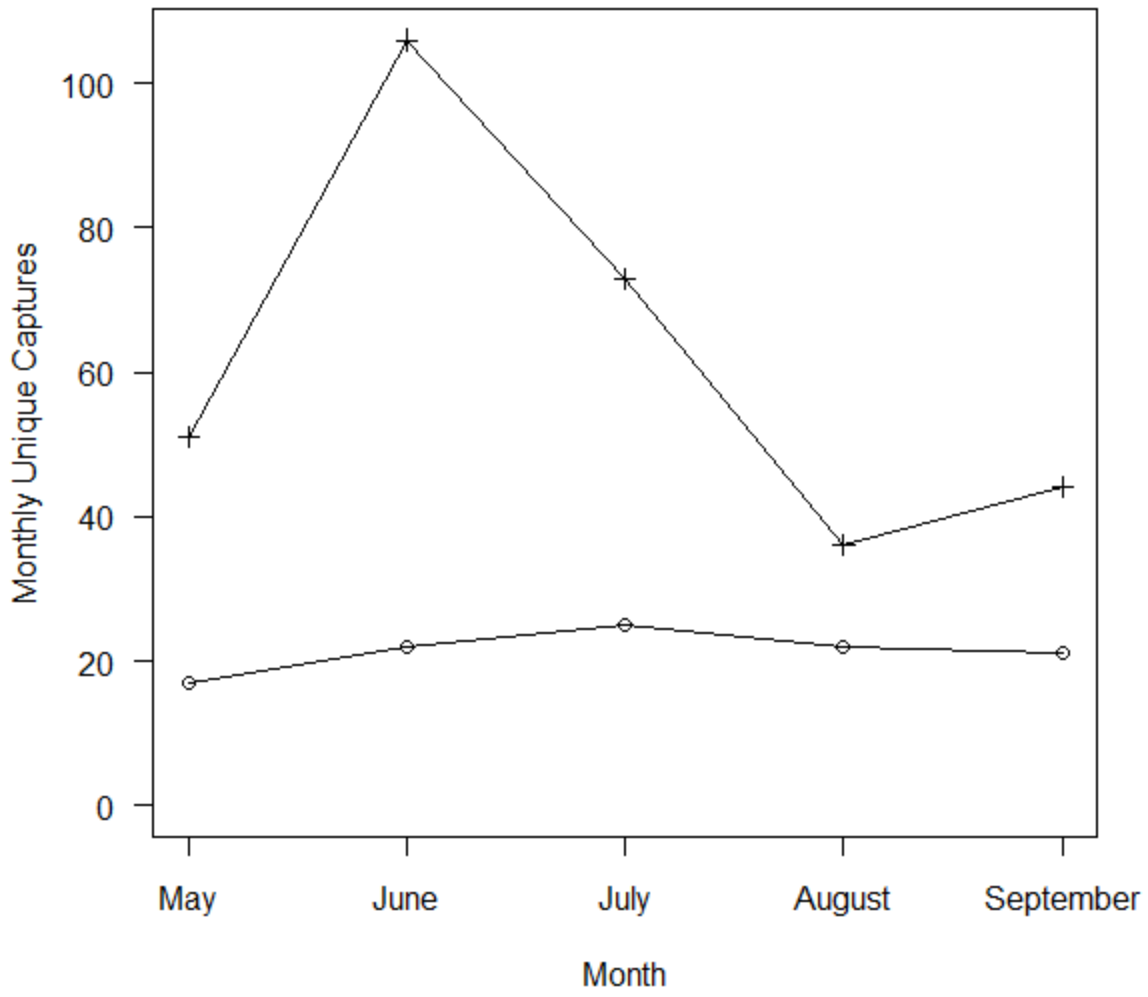


Figure 3. Monthly Unique Captures for *P. leucopus* (circles) *and T. striatus* (crosses) across the five month trapping period from May- September at Black Rock Forest in 2009. Monthly Unique Captures are defined here as the number of captures (of each species) that represent unique individuals (i.e. excluding animals recaptured during the study) to a given month.

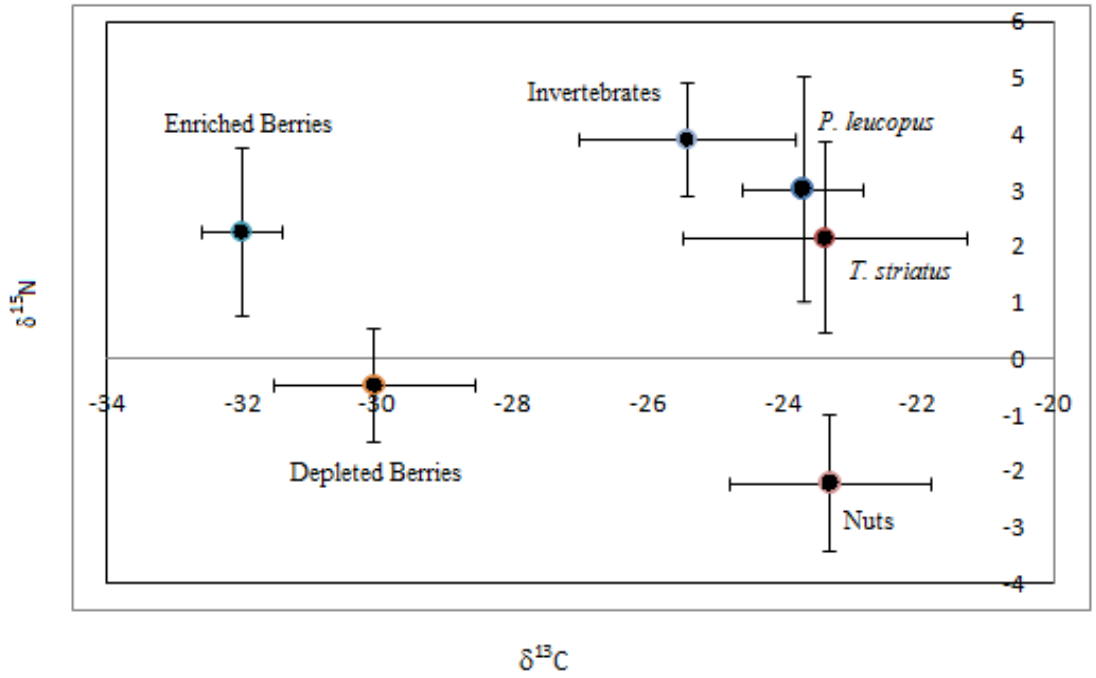


Figure 4. Stable isotopic ratios ($\delta^{13}C$, $\delta^{15}N$) for *P. leucopus*, *T. striatus*, and potential prey samples. Mean and standard error values are given. Carbon predator values were calculated by averaging both tissue and blood values, while nitrogen values were calculated by averaging tissue values.

Table 1. Total number of individual species captured across the trapping period (May-September 2009) at Black Rock Forest, New York. Mean home range (m^2) and captures per day are also given for each species (Home range citations given as subscripts).

Family	Species	Common Name	Monthly Captures						Home Range (m^2)
			M	J	Jul	A	S	Total	
Muridae	*Clethrionomys gapperi*	Southern red-backed vole	1	0	0	0	0	1	1.0-5.0[a]
	Microtus pinetorum	Woodland vole	0	0	1	3	2	6	700-2800[b]
	Peromyscus leucopus	White-footed mouse	75	85	89	78	78	405	1.0[c]
Mustelidae	*Mustela frenata*	Long-tailed weasel	0	0	0	0	1	1	51,000-180,000[d]
Sciuridae	*Tamias striatus*	Eastern chipmunk	182	384	307	142	119	1134	0.2-1.0[e]
	Glaucomys sabrinus	Northern flying squirrel	0	1	0	0	0	1	8.0-1.2[f]
Soricidae	*Sorex Cinereus*	Masked shrew	2	0	0	0	3	5	6.0[g]
	Blarina brevicauda	Northern short-tailed shrew	9	5	1	5	17	37	25.0[h]
	Sorex dispar	Long-Tailed shrew	0	0	0	1	0	1	1.0-4.0[i]
Capture Success*	n/a	n/a	1793.3	3173.3	2660	1526.7	1466.7	3540	n/a

* *Capture success is defined as the number of captures per 100 trapping days (captures/trapping days x 100)*

a. Merritt 1981
b. Kurta and Burt 1995
c. Lackey et al. 1985
d. Gehring and Swihart (2004)
e. Snyder 1982
f. Wells-Gosling and Heaney 1984
g. Nagorsen 1996
h. George et al. 1986
i. Kirkland and Hart 1999

Table 2. Number of A) *P. leucopus* and B) *T. striatus* captures by gender and age class across slope position (Lower, Mid, Upper) and trapping month (May- September) at Black Rock Forest, New York.

	Slope position			**Trapping Month**					
A	Lower	Mid	Upper	M	J	Jul	A	S	Total
Male	18	9	13	10	12	11	5	2	40
Female	10	11	4	7	10	4	2	2	25
Juvenile	11	0	9	7	6	2	2	3	20
sub-adult	6	10	4	3	8	8	1	0	20
Adult	11	11	3	7	6	2	6	4	25
B									
Male	14	31	33	29	23	20	4	2	78
Female	37	25	27	20	49	14	5	1	89
Juvenile	31	25	22	22	38	11	7	0	78
Adult	20	26	31	18	28	23	4	4	77

Table 3. Number of *P. leucopus* and *T. striatus* unique single plot (SP) and unique multi-plot (MP) captures across plot.

	P. leucopus		***T. striatus***	
Plot	**SP**	**MP**	**SP**	**MP**
a1	9	3	17	4
a2	3	3	7	6
a3	2	3	12	5
a4	4	3	4	3
b1	3	1	9	10
b2	2	2	8	5
b3	4	3	10	8
b4	3	3	13	6
c1	5	2	7	8
c2	2	0	12	8
c3	2	2	12	2
c4	5	2	9	6

Table 4. Percent relative frequency of occurrence (%RFO) of prey values found from small mammal scat from individuals captured at Black Rock Forest across four treatments, three slope positions, and five months (May to September) for A) *P. leucopus* and B) *T. striatus*

A	**%RFO Treatment**				**%RFO Slope position**			**%RFO Month**					
Prey Item	**C**	**N**	**O**	**O50**	**Lower**	**Mid**	**Upper**	**M**	**J**	**Jul**	**A**	**S**	**Total**
All Vegetation	63.6	53.7	59.1	52.0	55.8	50.0	60.0	47.6	56.7	62.1	50.0	55.6	55.6
Berries	9.1	4.9	0.0	0.0	2.3	0.0	6.7	0.0	6.7	3.4	0.0	0.0	3.0
Vaccinium[a]	9.1	4.9	0.0	0.0	2.3	0.0	6.7	0.0	6.7	3.4	0.0	0.0	3.0
Gaylussacia	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
R. phoenicolasius	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Other seed	0.0	7.3	4.5	4.0	9.3	0.0	3.3	0.0	3.3	13.8	0.0	0.0	5.1
Other vegetation	54.5	41.5	54.5	48.0	44.2	50.0	50.0	47.6	46.7	44.8	50.0	55.6	47.5
Invertebrates	36.4	46.3	40.9	48.0	44.2	50.0	40.0	52.4	43.3	37.9	50.0	44.4	44.4
Coleoptera	18.2	14.6	13.6	16.0	16.3	19.2	10.0	19.0	13.3	13.8	20.0	11.1	15.2
Hymenoptera	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Ixodida	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Trichoptera	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Diptera	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Other invertebrates	18.2	31.7	27.3	32.0	27.9	30.8	30.0	33.3	30.0	24.1	30.0	33.3	29.3
number of scats[b]	6	17	12	12	19	13	15	10	14	13	5	5	47
mean niche breadth[c]	0.2	0.1	0.1	0.06	0.1	0.2	0.1	0.2	0.2	0.2	0.2	0.1	0.1

B	**%RFO Treatment**				**%RFO Slope position**			**%RFO Month**					
Prey Item	**C**	**N**	**O**	**O50**	**Lower**	**Mid**	**Upper**	**M**	**J**	**Jul**	**A**	**S**	**Total**
Vegetation	51.8	51.0	51.5	51.0	58.0	50.0	45.3	39.6	45.9	64.6	55.2	44.4	51.3
Berries	6.0	3.8	3.0	6.3	7.6	4.4	2.3	0.0	3.7	8.0	13.8	0.0	4.9
Vaccinium[a]	2.4	1.9	3.0	2.1	3.8	1.1	1.6	0.0	2.8	2.7	6.9	0.0	2.3
Gaylussacia	2.4	1.9	0.0	1.0	3.1	0.0	0.8	0.0	0.9	2.7	3.4	0.0	1.4
R. phoenicolasius	1.2	0.0	0.0	3.1	0.8	3.3	0.0	0.0	0.0	2.7	3.4	0.0	1.1
Other seed	4.8	10.6	6.1	7.3	13.0	3.3	4.7	0.0	1.8	20.4	3.4	0.0	7.4
Other vegetation	41.0	36.5	42.4	37.5	37.4	42.2	38.3	39.6	40.4	36.3	37.9	44.4	39.0
Invertebrates	48.2	49.0	48.5	49.0	42.0	50.0	54.7	60.4	54.1	35.4	44.8	55.6	48.7
Coleoptera	14.5	15.4	9.1	10.4	9.9	10.0	17.2	18.7	13.8	8.0	6.9	11.1	12.6
Hymenoptera	4.8	1.9	3.0	6.3	3.8	5.6	3.1	5.5	5.5	2.7	0.0	0.0	4.0
Ixodida	0.0	2.9	1.5	1.0	1.5	1.1	1.6	2.2	0.0	1.8	3.4	0.0	1.4
Trichoptera	1.2	0.0	0.0	1.0	0.0	2.2	0.0	1.1	0.9	0.0	0.0	0.0	0.6
Diptera	1.2	1.0	0.0	2.1	1.5	1.1	0.8	0.0	0.9	1.8	3.4	0.0	1.1
Other invertebrates	26.5	27.9	34.8	28.1	25.2	30.0	32.0	33.0	33.0	21.2	31.0	44.4	28.9
number of scats[b]	34	38	28	36	49	38	49	36	44	41	11	4	136
mean niche breadth[c]	0.3	0.2	0.2	0.3	0.2	0.3	0.3	0.2	0.2	0.3	0.3	0.1	0.3

a. Vaccinium genus; unidentified species
b. Total number of fecal samples per specific treatment type/slope position/trapping month
c. Niche breadth determined from Levin's Measure (Krebs 1989)

Table 5. Tissue and blood stable carbon and nitrogen isotopic values across treatment type (C, N, O, O50), slope position (Lower, Mid, and Upper), and over time (Early, Mid-, and Late Trapping Season, abbreviated: ETS, MTS, and LTS, respectively) for A) *P. leucopus* and B) *T. striatus*. Mean values, standard error (SE), and number of samples (n) are given.

A	Blood $\delta^{13}C$		Blood $\delta^{15}N$				Tissue $\delta^{13}C$		Tissue $\delta^{15}N$		
Variable	mean	SE	mean	SE	N	Variable	mean	SE	mean	SE	n
Treatment C	-23.07	0.90	0.12	4.83	2	Treatment C	-23.85	0.32	1.98	2.67	7
Treatment N	-23.89	0.61	1.17	3.37	13	Treatment N	-24.16	0.77	3.24	2.19	9
Treatment O	-23.83	0.60	1.70	0.95	6	Treatment O	-23.24	1.65	2.97	2.84	5
Treatment O50	-23.64	n/a	3.08	n/a	1	Treatment O50	-22.75	1.36	4.47	1.86	4
Lower Slope	-23.76	0.59	1.43	3.17	12	Lower Slope	-23.80	1.09	3.35	2.55	13
Middle Slope	-23.98	0.74	-0.78	2.11	3	Mid-Slope	-23.10	1.45	3.45	2.86	5
Upper Slope	-23.74	0.72	1.98	2.43	7	Upper Slope	-23.81	0.81	2.14	1.90	7
ETS	-23.78	0.68	0.62	3.44	11	ETS	-23.65	1.03	2.78	2.64	14
MTS	-23.94	0.44	2.17	1.14	7	MTS	-24.14	0.65	3.02	1.90	8
LTS	-23.53	0.81	1.67	3.36	4	LTS	-22.44	1.66	4.24	3.07	3

B	Blood $\delta^{13}C$		Blood $\delta^{15}N$				Tissue $\delta^{13}C$		Tissue $\delta^{15}N$		
Variable	mean	SE	mean	SE	N	Variable	mean	SE	mean	SE	n
Treatment C	-23.56	0.54	0.21	2.37	11	Treatment C	-23.74	0.86	2.12	1.81	11
Treatment N	-23.65	0.40	0.78	2.39	13	Treatment N	-23.95	0.50	2.10	1.21	11
Treatment O	-23.46	0.65	-1.03	4.09	6	Treatment O	-23.23	1.14	2.99	2.52	6
Treatment O50	-23.57	0.38	0.82	3.00	10	Treatment O50	-23.83	0.44	1.83	1.51	11
Lower Slope	-23.70	0.51	1.35	2.40	17	Lower Slope	-23.63	0.79	2.65	1.59	9
Middle Slope	-23.49	0.33	-0.12	3.11	12	Mid-Slope	-23.73	0.85	2.45	1.83	17
Upper Slope	-23.48	0.52	-0.64	2.77	11	Upper Slope	-23.84	0.55	1.45	1.39	13
ETS	-23.44	0.47	-0.80	2.71	25	ETS	-23.74	0.79	2.00	1.78	31
MTS	-23.92	0.32	2.62	1.31	12	MTS	-23.94	0.35	2.82	1.13	6
LTS	-23.32	0.10	0.95	2.72	3	LTS	-23.22	0.56	2.79	1.18	2

Table 6. Percent contribution of potential prey sources (invertebrates, nuts, and vegetation) to A) *P. leucopus* and B) *T. striatus* diet across treatment type (C, N, O, O50). Mean ($\bar{x}$) values, standard error (SE), and number of potential source contributions (n) are given based on combined $\delta^{13}C$ tissue and blood and $\delta^{15}N$ tissue values and estimated from Isosource mixing models.

A	**invertebrate**		**nut**		**Vegetation**		
Variable	$\bar{x}$ **(%)**	**SE**	$\bar{x}$ **(%)**	**SE**	$\bar{x}$ **(%)**	**SE**	**n**
C	32.7	±1.6	50.3	±1.3	17.0	±1.6	24
N	54.3	±1.6	35.5	±1.4	10.2	±1.6	24
O	50.5	±1.7	43.5	±1.3	6.0	±1.6	22
O50	64.0	±2.3	34.7	±1.8	1.3	±1.2	23
B	**invertebrate**		**nut**		**Vegetation**		
Variable	$\bar{x}$ **(%)**	**SE**	$\bar{x}$ **(%)**	**SE**	$\bar{x}$ **(%)**	**SE**	**n**
C	37.7	±1.6	48.3	±1.3	14.0	±1.6	24
N	34.5	±1.6	47.8	±1.2	17.7	±1.7	23
O	51.5	±1.7	46.0	±1.3	2.5	±1.5	22
O50	30.0	±1.4	51.5	±1.5	18.5	±1.5	22

Table 7. Percent contribution of potential prey sources (invertebrates, nuts, and vegetation) to A) *P. leucopus* and B) *T. striatus* diet across slope position (Lower, Mid, and Upper) and over time (Early, Mid-, and Late Trapping Season, abbreviated below as ETS, MTS, and LTS). Mean ($\bar{x}$) values, standard error (SE), and number of potential source contributions (n) are given based on combined $\delta^{13}C$ tissue and blood and $\delta^{15}N$ tissue values and estimated from Isosource mixing models.

A	**invertebrate**		**nut**		**Vegetation**		
Variable	$\bar{x}$ **(%)**	**SE**	$\bar{x}$ **(%)**	**SE**	$\bar{x}$ **(%)**	**SE**	**n**
ALL	51.1	±1.6	41.0	±1.5	7.9	±1.6	24
Lower Slope	56.7	±2.0	37.3	±1.3	6.1	±1.6	23
Middle Slope	58.4	±1.6	40.3	±1.8	1.3	±1.2	23
Upper Slope	35.5	±1.6	47.7	±1.3	17.0	±1.6	22
ETS	46.9	±1.6	43.0	±1.5	10.1	±1.6	24
MTS	50.3	±1.6	36.7	±1.3	12.9	±1.6	23
LTS	63.5	±2.1	35.5	±1.8	1.0	±1.0	20
B	**invertebrate**		**nut**		**Vegetation**		
Variable	$\bar{x}$ **(%)**	**SE**	$\bar{x}$ **(%)**	**SE**	$\bar{x}$ **(%)**	**SE**	**n**
Total	35.3	±1.6	49.5	±1.4	15.3	±1.6	24
Lower Slope	44.5	±1.6	44.7	±1.3	10.8	±1.7	24
Middle Slope	41.1	±1.6	47.0	±1.5	11.9	±1.6	24
Upper Slope	23.2	±1.6	55.0	±1.3	21.8	±1.5	25
ETS	33.1	±1.6	51.3	±1.3	15.6	±1.5	21
MTS	46.9	±1.4	40.1	±1.4	13.0	±1.6	22
LTS	48.1	±1.6	48.7	±1.3	3.1	±1.5	21

Table 8. Mean air and soil temperature (°C) over the five-month trapping session (May-September 2009) at Black Rock Forest, New York.

	M		J		Jul		A		S	
Temperature	$\bar{x}$	SE	$\bar{x}$	SE	$\bar{x}$	SE	$\bar{x}$	SE	$\bar{x}$	SE
Soil (°C)	11.7	± 1.8	14.2	± 2.3	16.4	± 2.1	17.5	± 2.2	13.1	± 2.4
Air (°C)	14.4	± 5.3	18.1	± 2.0	20.2	± 1.8	20.6	± 3.1	15	± 3.0

Table 9. Number of invertebrates collected by order across each treatment type at Black Rock Forest, New York from May-September 2009.

Order	**C**	**N**	**O**	**O50**
Aranea	0	2	0	3
Chilopoda	0	0	0	6
Coleopteran	7	4	6	7
Dermaptera	2	3	2	1
Diplodopa	2	3	1	2
Diptera	5	9	3	5
hymenoptera	1	1	1	2
Lepidoptera	3	3	2	5
malacostraca	0	0	0	1
Unidentified	6	5	10	23
Total	26	30	25	55

Literature Cited

Abaturov, B. D. 1972. Role of burrowing animals in transport of mineral substances in soil. *Pedobiologia*, 12, 261.

Abbott, H. G. & Quink, T. F. 1970. Ecology of eastern white pine seed caches made by small forest mammals. *Ecology*, 51, 271-&.

Abramsky, Z. 1978. Small mammal community ecology: Changes in species-diversity in response to manipulated productivity. *Oecologia*, 34, 113-123.

Adamcikova, K., Kobza, M. & Juhasova, G. 2009. The development of population structure of *Cryphonectria parasitica* on European chestnut (*Castanea sativa*) in the experimental Castanetarium Horne Lefantovce, observed over a 12-year study period. *Horticultural Science*, 36, 55-60.

Alcantara, J. M., Rey, P. J., Sanchez-Lafuente, A. M. & Valera, F. 2000. Early effects of rodent post-dispersal seed predation on the outcome of the plant-seed disperser interaction. *Oikos*, 88, 362-370.

Andersen, D. C. & Macmahon, J. A. 1985. Plant succession following the Mount St. Helens volcanic-eruption: Facilitation by a burrowing rodent, *Thomomys talpoides*. *American Midland Naturalist*, 114, 62-69.

Angelstam, P., Lindstrom, E. & Widen, P. 1984. Role of predation in short-term population fluctuations of some birds and mammals in Fennoscandia. *Oecologia*, 62, 199-208.

Anthony, N. M., Ribic, C. A., Bautz, R. & Garland, T. 2005. Comparative effectiveness of Longworth and Sherman live traps. *Wildlife Society Bulletin*, 33, 1018-1026.

Apigian, K., Brown, L., Loda, J., Toas, S. & Allen-Diaz, B. 2005. Small mammal and herpetofaunal abundance and diversity along a gradient of sudden oak death infection. In: *General Technical Report*: Sudden Oak Death Second Science Symposium.

Baldwin, H. I. 1934. Some physiological effects of girdling northern hardwoods. *Bull. Torrey Bot. Club*, 61, 249-257.

Batzli, G. O. 1977. Population-dynamics of white-footed mouse in floodplain and upland forests. *American Midland Naturalist*, 97, 18-32.

Baxter, R. & Hansson, L. 2001. Bark consumption by small rodents in the northern and southern hemispheres. *Mammal Review*, 31, 47-59.

Bearhop, S. Adams, C., Waldron, S., Fuller, R., Macleod, H. 2004. Determining trophic niche width: a novel approach using stable isotope analysis. *Journal of Animal Ecology,* 73, 1007-1012.

Ben-David, M. & Schell, D. M. 2001. Mixing models in analyses of diet using multiple stable isotopes: a response. *Oecologia*, 127, 180-184.

Ben David, M., Bowyer, R. T. & Faro, J. B. 1996. Niche separation by mink and river otters: Coexistence in a marine environment. *Oikos*, 75, 41-48.

Bennett, P. M. & Hobson, K. A. 2009. Trophic structure of a boreal forest arthropod community revealed by stable isotope (delta C-13, delta N-15) analyses. *Entomological Science*, 12, 17-24.

Blackburn, G. A. & Milton, E. J. 1995. Seasonal-variations in the spectral reflectance of deciduous tree canopies. *International Journal of Remote Sensing*, 16, 709-720.

Bligh, E. G. & Dyer, W. J. 1959. A rapid method of total lipid extraction and purification. *Canadian Journal of Biochemistry and Physiology*, 37, 911-917.

Bond, W. J. & Breytenbach, G. J. 1985. Ants, rodents and seed predation in proteaceae. *South African Journal of Zoology*, 20, 150-154.

Boonstra, R. & Krebs, C. J. 2006. Population limitation of the northern red-backed vole in the boreal forests of northern Canada. *Journal of Animal Ecology*, 75, 1269-1284.

Brenner, F. J. & Lyle, P. D. 1975. Effect of previous photoperiodic conditions and visual stimulation of food storage and hibernation in the Eastern Chipmunks (*Tamias striatus*). *American Midland Naturalist,* 93, 227-234.

Brosofske, K. D., Chen, J. & Crow, T. R. 2001. Understory vegetation and site factors: Implications for a managed Wisconsin landscape. *Forest Ecology and Management*, 146, 75-87.

Brown, J. H. & Davidson, D. W. 1977. Competition between seed-eating rodents and ants in desert ecosystems. *Science*, 196, 880-882.

Brown, L. B. & Allen-Diaz, B. 2009. Forest stand dynamics and sudden oak death: Mortality in mixed-evergreen forests dominated by coast live oak. *Forest Ecology and Management*, 257, 1271-1280.

Buckner, C. H. 1966. Role of vertebrate predators in biological control of forest insects. *Annual Review of Entomology*, 11, 449-&.

Burt, W. H. 1940. Territorial behavior and populations of some small mammals in southern Michigan. *University of Michigan Museum of Zoology Miscellaneous Publications*, 1-58.

Carey, A. B. & Harrington, C. A. 2001. Small mammals in young forests: implications for management for sustainability. *Forest Ecology and Management*, 154, 289-309.

Carey, A. B. & Johnson, M. L. 1995. Small mammals in managed, naturally young, and old-growth forests. *Ecological Applications*, 5, 336-352.

Cazares, E., Luoma, D. L., Amaranthus, M. P., Chambers, C. L. & Lehmkuhl, J. F. 1999. Interaction of fungal sporocarp production with small mammal abundance and diet in Douglas-fir stands of the southern cascade range. *Northwest Science*, 73, 64-76.

Cazares, E. & Trappe, J. M. 1994. Spore dispersal of ectomycorrhizal fungi on a glacier forefront by mammal mycophagy. *Mycologia*, 86, 507-510.

Chambers, J. C. & Macmahon, J. A. 1994. A day in the life of a seed - movements and fates of seeds and their implications for natural and managed systems. *Annual Review of Ecology and Systematics*, 25, 263-292.

Chen, J., Saunders, S. C., Crow, T. R., Naiman, R. J., Brosofske, K. D., Mroz, G. D., Brookshire, B.L. & Franklin, J. F. 1999. Microclimate in forest ecosystem and landscape ecology. *Bioscence,* 49, 288-297.

Chew, R. M. 1978. The impact of small mammals on ecosystem structure and function. *Special Publications Pymatuning Laboratory of Ecology*, 167-180.

Churchfield, S., Hollier, J. & Brown, V. K. 1991. The effects of small mammal predators on grassland invertebrates, investigated by field exclosure experiment. *Oikos*, 60, 283-290.

Claridge, A. W., Tanton, M. T., Seebeck, J. H., Cork, S. J. & Cunningham, R. B. 1992. Establishment of ectomycorrhizae on the roots of 2 species of eucalyptus from fungal spores contained in the feces of the long-nosed potoroo (*Potorous tridactylus*). *Australian Journal of Ecology*, 17, 207-217.

Claridge, A. W., Trappe, J. M., Cork, S. J. & Claridge, D. L. 1999. Mycophagy by small mammals in the coniferous forests of North America: nutritional value of sporocarps of *Rhizopogon vinicolor*, a common hypogeous fungus. *Journal of Comparative Physiology B-Biochemical Systemic and Environmental Physiology*, 169, 172-178.

Clark, B. K. & Kaufman, D. W. 1991. Effects of plant litter on foraging and nesting behavior of prairie rodents. *Journal of Mammalogy*, 72, 502-512.

Clotfelter, E. D., Pedersen, A. B., Cranford, J. A., Ram, N., Snajdr, E. A., Nolan, V. & Ketterson, E. D. 2007. Acorn mast drives long-term dynamics of rodent and songbird populations. *Oecologia*, 154, 493-503.

Clough, G. C. 1987. Relations of small mammals to forest management in northern Maine. *Canadian Field-Naturalist*, 101, 40-48.

Connors, M. J., Schauber, E. M., Forbes, A., Jones, C. G., Goodwin, B. J. & Ostfeld, R. S. 2005. Use of track plates to quantify predation risk at small spatial scales. *Journal of Mammalogy*, 86, 991-996.

Cork, S. J. & Kenagy, G. J. 1989. Nutritional-value of hypogeous fungus for a forest-dwelling ground-squirrel. *Ecology*, 70, 577-586.

Cromer, R. B., Gresham, C. A., Goddard, M., Landham, J. D. & Hanlin, H. G. 2007. Associations between two bottomland hardwood forest shrew species and hurricane-generated woody debris. *Southeastern Naturalist*, 6, 235-246.

Darling, A. F. & Bayne, E. M. 2010. The potential of stable isotope (delta C-13, delta N-15) analyses for measuring foraging behaviour of animals in disturbed boreal forest. *Ecoscience*, 17, 73-82.

Davidson, J. M., Wickland, A. C., Patterson, H. A., Falk, K. R. & Rizzo, D. M. 2005. Transmission of Phytophthora ramorum in mixed-evergreen forest in California. *Phytopathology*, 95, 587-596.

DeNiro, M. J. & Epstein, S. 1978. Influence of diet on distribution of carbon isotopes in animals. *Geochimica Et Cosmochimica Acta*, 42, 495-506.

DeNiro, M. J. & Epstein, S. 1981. Influence of diet on the distribution of nitrogen isotopes in animals. *Geochimica Et Cosmochimica Acta*, 45, 341-351.

DeNiro, M. J. & Epstein, S. Influence of diet on the distribution of carbon isotopes in animals. *Geochemica et Cosmochemica Acta*, 42, 495-506.

Dizney, L., Jones, P. D. & Ruedas, L. A. Efficacy of three types of live traps used for surveying small mammals in the Pacific Northwest. *Northwestern Naturalist*, 89, 171-180.

Donahue, J. G., Piesman, J., Spielman, A. 1987. Reservoir competence of white-footed mice for Lyme disease spirochetes. *The American Society of Tropical Medicine and Hygiene,* 36, 92-96.

Drever, M. C. & Harestad, A. S. 1998. Diets of Norway Rats, *Rattus norvegicus*, on Langara Island, Queen Charlotte Islands, British Columbia: Implications for conservation of breeding seabirds. *Canadian Field-Naturalist*, 112, 676-683.

Dueser, R. D. & Shugart, H. H. 1978. Microhabitats in a forest-floor small mammal fauna. *Ecology*, 59, 89-98.

Dunford, C. 1972. Summer activity of eastern chipmunks. *Journal of Mammalogy*, 53, 176-&.

Elias, T. S. & Dykeman, P. 2009. *Edible Wild Plants: A North American Field Guide to over 200 Natural Foods*. New York: Sterling.

Elkinton, J. S., Healy, W. M., Buonaccorsi, J. P., Boettner, G. H., Hazzard, A. M., Smith, H. R. & Liebhold, A. M. 1996a. Interactions among gypsy moths, white-footed mice, and acorns. *Ecology*, 77, 2332-2342.

Elkinton, J. S., Liebhold, A. M. & Muzika, R. M. 2004. Effects of alternative prey on predation by small mammals on gypsy moth pupae. *Population Ecology*, 46, 171-178.

Elkinton, J. S., Healey, W. M., Buonaccorsi, J. P., Boettner, G. H., Hazzard, A. M. & Smith, H. R. 1996b. Interactions Among Gypsy Moths, White-footed Mice, and Acorns. *Ecological Society of America*, 77, 2332-2342.

Elliott, A. G. & Root, B. G. 2006. Small mammal responses to silvicultural and precipitation-related disturbance in northeastern Missouri riparian forests. *Wildlife Society Bulletin*, 34, 485-501.

Elliott, P. F. 1974. Evolutionary responses of plants to seed-eaters - pine squirrel predation on lodgepole pine. *Evolution*, 28, 221-231.

Erlinge, S. Feeding Habits of the Weasel *Mustela nivalis* in Relation to Prey Abundance. *Oikos*, 26, 378-384.

Evans, A. V. 2007. *Field Guide to Insects and Spiders of North America*. New York: Sterling Publishing Co.

Farquhar, G. D., Ehleringer, J. R. & Hubick, K. T. 1989. Carbon isotope discrimination and photosynthesis. *Annual Review of Plant Physiology and Plant Molecular Biology*, 40, 503-537.

Fichtner, E. J., Lynch, S. C. & Rizzo, D. M. 2007. Detection, distribution, sporulation, and survival of *Phytophthora ramorum* in a California redwood-tanoak forest soil. *Phytopathology*, 97, 1366-1375.

Flaherty, E. A., Ben-David, M. & Smith, W. P. 2010. Diet and food availability: implications for foraging and dispersal of Prince of Wales northern flying squirrels across managed landscapes. *Journal of Mammalogy*, 91, 79-91.

Fogel, R. & Trappe, J. M. 1978. Fungus consumption (mycophagy) by small animals. *Northwest Science*, 52, 1-31.

Ford, A. T. & Fahrig, L. 2008. Movement patterns of eastern chipmunks (*Tamias striatus*) near roads. *Journal of Mammalogy*, 89, 895-903.

France, R. 1996. Carbon isotope ratios in logged and unlogged boreal forests: Examination of the potential for determining wildlife habitat use. *Environmental Management*, 20, 249-255.

Frank, J. L., Barry, S. & Southworth, D. 2006. Mammal mycophagy and dispersal of mycorrhizal inoculum in Oregon white oak woodlands. *Northwest Science*, 80, 264-273.

Fraser, L. H. & Madson, E. B. 2008. The interacting effects of herbivore exclosures and seed addition in a wet meadow. *Oikos*, 117, 1057-1063.

Freeman, A. 2010. The Effect of Coarse Woody Debris on Species Diversity in the Boreal Forest, University of Waterloo.

Fryxell, J. M., Falls, J. B., Falls, E. A. & Brooks, R. J. 1998. Long-term dynamics of small-mammal populations in Ontario. *Ecology*, 79, 213-225.

Garcia, D., Obeso, J. R. & Martinez, I. 2005. Rodent seed predation promotes bird-dispersed trees in temperate differential recruitment among secondary forests. *Oecologia*, 144, 435-446.

Gehring, T. M. & Swihart, R. K. 2004. Home range and movements of long-tailed weasels in a landscape fragmented by agriculture. *Journal of Mammalogy*, 85, 79-86.

Geiger, R. 1965. *The Climate Near the Ground*. Cambridge, MA: Harvard University Press.

Gehlhausen, S. M., Schwartz, M. W., Augspurger, C. K. 2000. Vegetation and microclimate edge effects in two mixed-mesophytic forest fragments. *Plant Ecology*, 147, 21-35.

George, S. B., Choate, J. R. & Genoways, H. H. 1986. *Blarina brevicauda*. *Mammalian Species*, 1-9.

Gibbs, G. W. 2009. The end of an 80-million year experiment: a review of evidence describing the impact of introduced rodents on New Zealand's 'mammal-free' invertebrate fauna. *Biological Invasions*, 11, 1587-1593.

Gordon, R. B. & Malone, P. M. 1997. *The texture of Industry: An Archaelogical View of the Industralization of North America*. New York: Oxford University Press.

Gravel, D., Beaudet, M. & Messier, C. Large-scale synchrony of gap dynamics and the distribution of understory tree species in maple-beech forests. *Oecologia*, 162, 153-161.

Greenberg, C. H. 2002. Response of white-footed mice (*Peromyscus leucopus*) to coarse woody debris and microsite use in southern Appalachian treefall gaps. *Forest Ecology and Management*, 164, 57-66.

Hafner, M. S. 1977. Density and diversity in Mojave desert rodent and shrub communities. *Journal of Animal Ecology*, 46, 925-938.

Hanski, I., Turchin, P., Korpimaki, E. & Henttonen, H. 1993. Population oscillations of boreal rodents - regulation by mustelid predators leads to chaos. *Nature*, 364, 232-235.

Hanski, I., Hansson, L. & Henttonen, H. 1991. Specialist Predators, Generalist Predators, and the Microtine Rodent Cycle. *British Ecological Society*, 60, 353-367.

Hansson, L. A., Gyllstrom, M., Stahl-Delbanco, A. & Svensson, M. 2004. Responses to fish predation and nutrients by plankton at different levels of taxonomic resolution. *Freshwater Biology*, 49, 1538-1550.

Heithaus, E. R. 1981. Seed predation by rodents on 3 ant-dispersed plants. *Ecology*, 62, 136-145.

Henttonen, H., McGuire, A. D. & Hansson, L. 1985. Comparisons of amplitudes and frequencies (spectral analyses) of density variations in long-term data sets of *clethrionomys* species. *Annales Zoologici Fennici*, 22, 221-227.

Hilderbrand, G. V., Farley, S. D., Robbins, C. T., Hanley, T. A., Titus, K. & Servheen, C. 1996. Use of stable isotopes to determine diets of living and extinct bears. *Canadian Journal of Zoology-Revue Canadienne De Zoologie*, 74, 2080-2088.

Hobson, K.A. and R.G. Clark. 1992. Assessing avian diets using stable isotopes II: Factors influencing diet-tissue fractionation. *The Condor*, 94, 189-197.

Hobson, K.A. 1999. Stable-Carbon and Nitrogen Isotope Ratios of Songbird Feathers Grown in Two Terrestrial Biomes: Implications for Evaluating Trophic Relationships and Breeding Origins. *The Condor*, 101, 799-805.

Hogberg, P., Johannisson, C., Hogberg, M., Hogbom, L., Nasholm, T. & Hallgren, J. E. 1995. Measurements of abundances of n-15 and c-13 as tools in retrospective studies of n balances and water-stress in forests - a discussion of preliminary-results. *Plant and Soil*, 168, 125-133.

Homer, E. B., Taylor, J. M. & Padykula, H. A. 1964. Food habits and gastric morphology of the grasshopper mouse. *Journal of Mammalogy.*, 45, 513-535.

Horn, S. & Hanula, J. L. 2008. Relationship of coarse woody debris to arthropod availability for red-cockaded woodpeckers and other bark-foraging birds on loblolly pine boles. *Journal of Entomological Science*, 43, 153-168.

Howard, W. E., Marsh, R. E. & Cole, R. E. 1968. Food detection by deer mice using olfactory rather than visual cues. *Animal Behaviour*, 16, 13.

Hulme, P. E. 1998. Post-dispersal seed predation and seed bank persistence. *Seed Science Research*, 8, 513-519.

Humphries, M. M. & Boutin, S. 1999. Mass-dependent reproduction or reproduction-dependent mass? A comment on body mass and first time reproduction in female sciurids. *Canadian Journal of Zoology*, 77, 171-173.

Huntly, N. & Inouye, R. 1988. Pocket gophers in ecosystems - patterns and mechanisms. *Bioscience*, 38, 786-793.

Hurlbert, S. H. 1978. Measurement of niche overlap and some relatives. *Ecology*, 59, 67-77.

Hwang, Y. T., Millar, J. S. & Longstaffe, F. J. 2007. Do delta 15 N and delta 13C values of feces reflect the isotopic composition of diets in small mammals? *Canadian Journal of Zoology*, 85, 388-396.

Inouye, R. S., Huntly, N. J., Tilman, D. & Tester, J. R. 1987. Pocket gophers (*geomys-bursarius*), vegetation, and soil-nitrogen along a successional sere in east central Minnesota. *Oecologia*, 72, 178-184.

Jabin, M., Mohr, D., Kappes, H. & Topp, W. 2004. Influence of deadwood on density of soil macro-arthropods in a managed oak-beech forest. *Forest Ecology and Management*, 194, 61-69.

Jacobs, D. F., Selig, M. F. & Severeid, L. R. 2009. Aboveground carbon biomass of plantation-grown American chestnut (*Castanea dentata*) in absence of blight. *Forest Ecology and Management*, 258, 288-294.

Jacobs, J. M., Spence, J. R. & Langor, D. W. 2007. Influence of boreal forest succession and dead wood qualities on saproxylic beetles. *Agricultural and Forest Entomology*, 9, 3-16.

Janzen, D. H. 1982. Citation classic - seed predation by animals. *Current Contents/Agriculture Biology & Environmental Sciences*, 18-18.

Johnson, C. N. 1996. Interactions between mammals and ectomycorrhizal fungi. *Trends in Ecology & Evolution*, 11, 503-507.

Johnson, M. D. 1975. Seasonal and microseral variations in insect populations on carrion. *American Midland Naturalist*, 93, 79-90.

Johnson, R., Ferguson, J. W. H., van Jaarsveld, A. S., Bronner, G. N. & Chimimba, C. T. 2002. Delayed responses of small-mammal assemblages subject to afforestation-induced grassland fragmentation. *Journal of Mammalogy*, 83, 290-300.

Kavanau, J. L. 1969. Influences of light on activity of small mammals. *Ecological Society of America*, 50, 548-557.

Kelt, D. A., Meserve, P. L., Nabors, L. K., Forister, M. L. & Gutierrez, J. R. 2004. Foraging ecology of small mammals in semiarid Chile: The interplay of biotic and abiotic effects. *Ecology*, 85, 383-397.

Kesner, M. H. & Linzey, A. V. 1997. Modeling population variation in *Peromyscus leucopus*: An exploratory analysis. *Journal of Mammalogy*, 78, 643-654.

Kirkland, G. L. 1990. Patterns of initial small mammal community change after clear cutting of temperate north-American forests. *OIKOS,* 59, 313-320.

Kirkland, G. L. & Hart, J. A. 1999. Recent distributional records for ten species of small mammals in Pennsylvania. *Northeastern Naturalist*, 6, 1-18.

Kirkland, G. L. & Layne, J. N. 1989. *Advances in the study of Peromyscus (Rodentia)*. Lubbock, TX: Texas Tech University Press.

Kjellander, P. & Nordstrom, J. 2003. Cyclic voles, prey switching in red fox, and roe deer dynamics - a test of the alternative prey hypothesis. *Oikos*, 101, 338-344.

Koenig, W. D., Schaefer, D. J., Mambelli, S. & Dawson, T. E. 2008. Acorns, insects, and the diet of adult versus nestling Acorn Woodpeckers. *Journal of Field Ornithology*, 79, 280-285.

Korpimaki, E. & Krebs, C. J. 1996. Predation and population cycles of small mammals - A reassessment of the predation hypothesis. *Bioscience*, 46, 754-764.

Korpimaki, E., Norrdahl, K., Huitu, O. & Klemola, T. 2005. Predator-induced synchrony in population oscillations of coexisting small mammal species. *Proceedings of the Royal Society B-Biological Sciences*, 272, 193-202.

Krebs, C. 1998. Ecological Methodology. New York: Addison Wesley Longman, Inc.

Kritzman, E. B. 1974. Ecological relationships of *Peromyscus maniculatus* and *perognathus-parvus* in eastern Washington. *Journal of Mammalogy*, 55, 172-188.

Kruuk, H. & Parish, T. 1981. Feeding specialization of the European badger *Meles meles* in Scotland. *Journal of Animal Ecology*, 50, 773-788.

Kurta, A. & Burt, W. H. 1995. *Mammals of the Great Lakes Region*. Ann Arbor, MI: The University of Michigan Press.

Lack, D. 1954. Cyclic mortality. *Journal of Wildlife Management*, 18, 25-37.

Lackey, J. A., Huckaby, D. G. & Ormiston, B. G. 1985. *Peromyscus leucopus*. *Mammalian Species*, 1-10.

Lacki, M. J., Gregory, M. J. & Williams, P. K. 1984. Summer activity of *Tamias striatus* in response to supplemented food. *Journal of Mammalogy*, 65, 521-524.

Landry-Cuerrier, M., Munro, D., Thomas, D. W. & Humphries, M. M. 2008. Climate and resource determinants of fundamental and realized metabolic niches of hibernating chipmunks. *Ecological Society of America,* 89, 3306-3316.

Larkin, J. L., Maehr, D. S., Krupa, J. J., Cox, J. J., Alexy, K., Unger, D. E. & Barton, C. 2008. Small Mammal Response to Vegetation and Spoil Conditions on a Reclaimed Surface Mine in Eastern Kentucky. *Southeastern Naturalist*, 7, 401-412.

Laundre, J. W. 1993. Effects of small mammal burrows on water infiltration in a cool desert environment. *Oecologia*, 94, 43-48.

Laundre, J. W. & Reynolds, T. D. 1993. Effects of soil-structure on burrow characteristics of 5 small mammal species. *Great Basin Naturalist*, 53, 358-366.

Loeb, S. C. 1993. The Role of Coarse Woody Debris in the Ecology of Southeastern Mammals. In: *Proceedings of the Workshop on Coarse Woody Debris in Southern Forests: Effects on Biodiversity* (Ed. by D. A. Crossley & J. W. McMinn). Athens, GA: Southern Research Station.

Loew, S. S. 1999. Sex-biased dispersal in eastern chipmunks, *Tamias striatus*. *Biomedical and Life Sciences*, 13, 557-577.

LoGiudice, K. & Ostfeld, R. S. 2002. Interactions between mammals and trees: predation on mammal-dispersed seeds and the effect of ambient food. *Oecologia*, 130, 420-425.

Loo, J. 2009. Ecological impacts of non-indigenous invasive fungi as forest pathogens. *Biological Invasions*, 11, 81-96.

Loo, J. A., Beardmore, T. L., Simpson, J. D. & McPhee, D. A. 2007. Tree species of concern in New Brunswick, Canada. I. Current status and threats. *Forestry Chronicle*, 83, 393-401.

Lorrain, A., Graham, B., Menard, F., Popp, B., Bouillon, S., van Breugel, P. & Cherel, Y. 2009. Nitrogen and carbon isotope values of individual amino acids: a tool to study foraging ecology of penguins in the Southern Ocean. *Marine Ecology-Progress Series*, 391, 293-306.

Lovett, G. M., Canham, C. D., Arthur, M. A., Weathers, K. C. & Fitzhugh, R. D. 2006. Forest ecosystem responses to exotic pests and pathogens in eastern North America. *Bioscience*, 56, 395-405.

Mackay, R. J. & Kalff, J. 1969. Seasonal variation in standing crop and species diversity of insect communities in a small Quebec stream. *Ecology*, 50, 101-&.

Martell, A. M. 1983. Changes in small mammal communities after logging in north-central Ontario. *Canadian Journal of Zoology-Revue Canadienne De Zoologie*, 61, 970-980.

Maser, C., Nussbaum, R. A. & Trappe, J. M. 1978. Fungal small mammal interrelationships with emphasis on Oregon coniferous forests. *Ecology*, 59, 799-809.

Matthews, R. W. & Matthews, J. R. 1970. Malaise trap studies of flying insects in a New York mesic forest. 1. Ordinal composition and seasonal abundance. *Journal of the New York Entomological Society*, 78, 52-&.

McCain, C. M. 2005. Elevational gradients in diversity of small mammals. *Ecology*, 86, 366-372.

McCay, T. S. & Komoroski, M. J. 2004. Demographic responses of shrews to removal of coarse woody debris in a managed pine forest. *Forest Ecology and Management*, 189, 387-395.

McFadden, K. W., Sambrotto, R. N., Medellin, R. A. & Gompper, M. E. 2006. Feeding habits of endangered pygmy raccoons (*Procyon pygmaeus*) based on stable isotope and fecal analyses. *Journal of Mammalogy*, 87, 501-509.

McKelvey, K. S. & Pearson, D. E. 2001. Population estimation with sparse data: the role of estimators versus indices revisited. *Canadian Journal of Zoology-Revue Canadienne De Zoologie*, 79, 1754-1765.

McLoskey, R. T. & Fieldwick, B. 1975. Ecological separation of sympatric rodents (P*eromyscus and Microtus*). *Journal of Mammalogy*, 56, 119-129.

McShea, W. J. 2000. The influence of acorn crops on annual variation in rodent and bird populations. *Ecology*, 81, 228-238.

McShea, W. J., Healy, W. M., Devers, P., Fearer, T., Koch, F. H., Stauffer, D. & Waldon, J. 2007. Forestry matters: Decline of oaks will impact wildlife in hardwood forests. *Journal of Wildlife Management*, 71, 1717-1728.

Meentemeyer, R. K., Rank, N. E., Anacker, B. L., Rizzo, D. M. & Cushman, J. H. 2008. Influence of land-cover change on the spread of an invasive forest pathogen. *Ecological Applications*, 18, 159-171.

Meentemeyer, R., Rizzo, D., Mark, W. & Lotz, E. 2004. Mapping the risk of establishment and spread of sudden oak death in California. *Forest Ecology and Management*, 200, 195-214.

Menzel, M. A., Ford, W. M., Laerm, J. & Krishon, D. 1999. Forest to wildlife opening: habitat gradient analysis among small mammals in the southern Appalachians. *Forest Ecology and Management*, 114, 227-232.

Merritt, J. F. 1981. *Clethrionomys gapperi*. *Mammalian Species*, 1-9.

Miller, D. H. & Getz, L. L. 1977a. Comparisons of population-dynamics of *Peromyscus* and *Clethrionomys* in New England. *Journal of Mammalogy*, 58, 1-16.

Miller, D. H. & Getz, L. L. 1977b. Factors influencing local distribution and species-diversity of forest small mammals in New England. *Canadian Journal of Zoology-Revue Canadienne De Zoologie*, 55, 806-814.

Miller, J. F., Millar, J. S. & Longstaffe, F. J. Carbon- and nitrogen-isotope tissue-diet discrimination and turnover rates in deer mice, *Peromyscus maniculatus*. *Canadian Journal of Zoology*, 86, 685.

Minagawa, M. & Wada, E. 1984. Stepwise enrichment of N-15 along food-chains - further evidence and the relation between delta-n-15 and animal age. *Geochimica Et Cosmochimica Acta*, 48, 1135-1140.

Monthey, R. W. & Soutiere, E. C. 1985. Responses of small mammals to forest harvesting in northern Maine. *Canadian Field-Naturalist*, 99, 13-18.

Moore, J. E. & Swihart, R. K. 2008. Factors affecting the relationship between seed removal and seed mortality. *Canadian Journal of Zoology-Revue Canadienne De Zoologie*, 86, 378-385.

Moores, A. R., Seymour, R. S. & Kenefic, L. S. 2007. Height development of shade-tolerant conifer saplings in multi-aged Acadian forest stands. *Canadian Journal of Forest Research-Revue Canadienne De Recherche Forestiere*, 37, 2715-2723.

Morin, R. S., Liebhold, A. M., Tobin, P. C., Gottschalk, K. W. & Luzader, E. 2007. Spread of beech bark disease in the eastern United States and its relationship to regional forest composition. *Canadian Journal of Forest Research-Revue Canadienne De Recherche Forestiere*, 37, 726-736.

Moritz, C., Patton, J. L., Conroy, C. J., Parra, J. L., White, G. C. & Beissinger, S. R. 2008. Impact of a century of climate change on small-mammal communities in Yosemite National Park, USA. *Science*, 322, 261-264.

Mountford, E. P., Savill, P. S. & Bebber, D. P. 2006. Patterns of regeneration and ground vegetation associated with canopy gaps in a managed beechwood in southern England. *Forestry*, 79, 389-408.

Nagorsen, D. W. 1996. *Opossums, shrews, and moles of British Columbia*. Vancouver: UBC Press.

Nardoto, G. B., Ferraz, P. D. G., de Barros, E. S., Ometto, J. & Martinelli, L. A. 2006. Stable carbon and nitrogen isotopic fractionation between diet and swine tissues. *Scientia Agricola*, 63, 579-582.

Nicolas, V. & Colyn, M. 2006. Relative efficiency of three types of small mammal traps in an African rainforest. *Belgian Journal of Zoology*, 136, 107-111.

Noel, A. R. A. 1970. The girdled tree. *Botanical Review*, 36, 162-195.

Nordyke, K. A. & Buskirk, S. W. 1991. Southern red-backed vole, *Clethrionomys gapperi*, populations in relation to stand succession and old-growth character in the central rocky-mountains. *Canadian Field-Naturalist*, 105, 330-334.

Norrdahl, K. & Korpimaki, E. 2000. Do predators limit the abundance of alternative prey? Experiments with vole-eating avian and mammalian predators. *Oikos*, 91, 528-540.

Ofarrell, T. P. 1975. Seasonal and altitudinal variations in populations of small mammals on rattlesnake mountain, Washington. *American Midland Naturalist*, 94, 190-204.

Ogilvie, D. M. & Stinson, R. H. 1966. Temperature selection in *Peromyscus* and laboratory mice, *Mus musculus*. *Journal of Mammalogy*, 47, 655.

Oliver, P. M. 1972. *Edible Wild Plants*: Macmillan Publishing Company.

Ostfeld, R. S., Manson, R. H. & Canham, C. D. 1997. Effects of rodents on survival of tree seeds and seedlings invading old fields. *Ecology*, 78, 1531-1542.

Ostry, M. E. & Moore, M. 2007. Natural and experimental host range of *Sirococcus clavigignenti-juglandacearum*. *Plant Disease*, 91, 581-584.

Oswald, C. D. & Flake, L. D. 1994. Bait Formulation Effectiveness in Live-Trapping Small Mammals in Eastern South Dakota. *Proceedings of the South Dakota Academy of Science*, 73, 101-108.

Owen, J. G. 1988. On productivity as a predictor of rodent and carnivore diversity. *Ecology*, 69, 1161-1165.

Panuska, J. A. & Wade, N. J. 1956. The burrow of *Tamias striatus*. *Journal of Mammalogy*, 37, 22-31.

Pardini, R., de Souza, S. M., Braga-Neto, R. & Metzger, J. P. 2005. The role of forest structure, fragment size and corridors in maintaining small mammal abundance and diversity in an Atlantic forest landscape. *Biological Conservation*, 124, 253-266.

Parmenter, C. A., Yates, T. L., Parmenter, R. R. & Dunnum, J. L. 1999. Statistical sensitivity for detection of spatial and temporal patterns in rodent population densities. *Emerging Infectious Diseases*, 5, 118-125.

Parmenter, R. R., Yates, T. L., Anderson, D. R., Burnham, K. P., Dunnum, J. L., Franklin, A. B., Friggens, M. T., Lubow, B. C., Miller, M., Olson, G. S., Parmenter, C. A., Pollard, J., Rexstad, E., Shenk, T. M., Stanley, T. R. & White, G. C. 2003. Small-mammal density estimation: A field comparison of grid-based vs. web-based density estimators. *Ecological Monographs*, 73, 1-26.

Pavlik, L. L., Pakhotin, P. I. & Moshkov, D. A. 1991. The influence of long hypothermia on the structure and function of hippocampal surviving slices from the guinea-pig and ground-squirrel. *Tsitologiya*, 33, 23-&.

Pearce, J. & Venier, L. 2005. Small mammals as bioindicators of sustainable boreal forest management. *Forest Ecology and Management*, 208, 153-175.

Phillips, D. L. & Gregg, J. W. 2003. Source partitioning using stable isotopes: coping with too many sources. *Oecologia*, 136, 261-269.

Phillips, D. L., Newsome, S. D. & Gregg, J. W. 2005. Combining sources in stable isotope mixing models: alternative methods. *Oecologia*, 144, 520-527.

Pyare, S. & Longland, W. S. 2001. Patterns of ectomycorrhizal-fungi consumption by small mammals in remnant old-growth forests of the Sierra Nevada. *Journal of Mammalogy*, 82, 681-689.

Quillfeldt, P., Schenk, I., McGill, R. A. R., Strange, I. J., Masello, J. F., Gladbach, A., Roesch, V. & Furness, R. W. 2008. Introduced mammals coexist with seabirds at New Island, Falkland Islands: abundance, habitat preferences, and stable isotope analysis of diet. *Polar Biology*, 31, 333-349.

Rhodes, D. H. & Richmond, M. E. 1985. Influence of soil texture, moisture and temperature on nest-site selection and burrowing by the pine vole, microtus pinetorum. *American Midland Naturalist*, 113, 102-108.

Rizzo, D. M. & Garbelotto, M. 2003. Sudden oak death: endangering California and Oregon forest ecosystems. *Frontiers in Ecology and the Environment*, 1, 197-204.

Santos, T. & Telleria, J. L. 1997. Vertebrate predation on Holm Oak, Quercus ilex, acorns in a fragmented habitat: effects on seedling recruitment. *Forest Ecology and Management*, 98, 181-187.

Sare, D. T. J., Millar, J. S. & Longstaffe, F. J. 2005. Tracing dietary protein in red-backed voles (Clethrionomys gapperi) using stable isotopes of nitrogen and carbon. *Canadian Journal of Zoology-Revue Canadienne De Zoologie*, 83, 717-725.

Savely, H. E. J. 1939. Ecological relations of certain animals in dead pine and oak logs. *Ecological Monographs*, 9, 321-385.

Schauber, E. M., Ostfeld, R. S. & Evans, A. S. 2005. What is the best predictor of annual Lyme disease incidence: Weather, mice, or acorns? *Ecological Applications*, 15, 575-586.

Schmidt-Holmes, S. & Drickamer, L. C. 2001. Impact of forest patch characteristics on small mammal communities: a multivariate approach. *Biological Conservation*, 99, 293-305.

Schnurr, J. L., Canham, C. D., Ostfeld, R. S. & Inouye, R. S. 2004. Neighborhood analyses of small-mammal dynamics: Impacts on seed predation and seedling establishment. *Ecology*, 85, 741-755.

Schnurr, J. L., Ostfeld, R. S. & Canham, C. D. 2002. Direct and indirect effects of masting on rodent populations and tree seed survival. *Oikos*, 96, 402-410.

Schoeninger, M. J. & Deniro, M. J. 1984. Nitrogen and carbon isotopic composition of bone-collagen from marine and terrestrial animals. *Geochimica Et Cosmochimica Acta*, 48, 625-639.

Schooley, J. P. 1934. Early development in some Sciuridae. *Journal of Morphology*, 56, 477-511.

Sealander, J. A., Griffin, D. N., Decosta, J. J. & Jester, D. B. 1958. A technique for studying behavioral-responses of small mammals to traps. *Ecology*, 39, 541-542.

Seto, S. 2009. Small Mammal Response to Oak Loss, NY. Columbia University.

Shaner, P. J., Bowers, M. & Macko, S. 2007. Giving-up density and dietary shifts in the white-footed mouse, *Peromyscus leucopus*. *Ecology*, 88, 87-95.

Shirley, H. L. 1929. The influence of light intensity and light quality upon the growth and survival of plants. *American Journal of Botany,* 16, 354-390.

Smith, M. H., Gentry, J. B. & Pinder, J. 1974. Annual fluctuations in small mammal population in an eastern hardwood forest. *Journal of Mammalogy*, 55, 231-234.

Snyder, D. P. 1982. *Tamias striatus*. *Mammalian Species*, 168, 1-8.

Sponheimer, M., Robinson, T. F., Roeder, B. L., Passey, B. H., Ayliffe, L. K., Cerling, T. E., Dearing, M. D. & Ehleringer, J. R. 2003. An experimental study of nitrogen flux in llamas: is N-14 preferentially excreted? *Journal of Archaeological Science*, 30, 1649-1655.

Stapp, P. 2002. Stable isotopes reveal evidence of predation by ship rats on seabirds on the Shiant Islands, Scotland. *Journal of Applied Ecology*, 39, 831-840.

Stephens, D. W. & Krebs, J. R. 1986. *Foraging Theory*. Princeton: Princeton University Press.

Stevens, V. 1997. The Ecological Role of Coarse Woody Debris: An Overview of the Ecological Importance of Coarse Woody Debris in British Columbia Forests, Ministry of Forests Research Program.

Stinson, N. J. 1978. Habitat structure and rodent species diversity on north and south-facing slopes in the Colorado lower montane zone. *The Southwestern Naturalist*, 23.

Storer, A. J., Rosemier, J. N., Beachy, B. L. & Flaspohler, D. J. In: *Beech Bark Disease: Proceedings of the Beech Bark Disease Symposium* (Ed. by C. A. Evans, J. A. Lucas & M. J. Twery), pp. 72-78. Newtown Square, PA: US. Department of Agriculture.

Strandga.H. 1967. Reliability of Petersen method tested on a roe-deer population. *Journal of Wildlife Management*, 31, 643.

Sullivan, T. P. 1979. Use of alternative foods to reduce conifer seed predation by the deer mouse, (*Peromyscus maniculatus*). *Journal of Applied Ecology*, 16, 475-495.

Sullivan, T. P., Lautenschlager, R. A. & Wagner, R. G. 1999. Clearcutting and burning of northern spruce-fir forests: implications for small mammal communities. *Journal of Applied Ecology*, 36, 327-344.

Tempel, D. J., Tietje, W. D. & Winslow, D. E. 2005. Vegetation and small vertebrates of oak woodlands at low and high risk for sudden oak death in San Luis Obispo County, California. In: *General Technical Report*, pp. 211-232: Sudden Oak Death Second Science Symposium.

Terman, C. R. 1966. Population fluctuations of *Peromyscus maniculatus* and other small mammals as revealed by north American census of small mammals. *American Midland Naturalist*, 76, 419.

Black Rock Forest Consortium. 2006. What Will Happen to Our Forests if the Oaks Die? *Black Rock Forest News*, 16, 4-5.

Tieszen, L. L., Boutton, T. W., Tesdahl, K. G. & Slade, N. A. 1983. Fractionation and turnover of stable carbon isotopes in animal-tissues - implications for delta C-13 analysis of diet. *Oecologia*, 57, 32-37.

Tietje, W. D., Lee, D. E. & Vreeland, J. K. 2008. Survival and abundance of three species of mice in relation to density of shrubs and prescribed fire in understory of an oak woodland in California. *Southwestern Naturalist*, 53, 357-369.

Tietje, W. D. & Vreeland, J. K. 1997. Molt of northern shovelers wintering in south Texas. *Southwestern Naturalist*, 42, 454-459.

Trappe, J. M. & Maser, C. 1976. Germination of spores of *Glomus macrocarpus* (endogonaceae) after passage through a rodent digestive-tract. *Mycologia*, 68, 433-436.

Trudell, S. A., Rygiewicz, P. T. & Edmonds, R. L. 2004. Patterns of nitrogen and carbon stable isotope ratios in macrofungi, plants and soils in two old-growth conifer forests. *New Phytologist*, 164, 317-335.

Turchin, P. 1990. Rarity of density dependence or population regulation with lags. *Nature*, 344, 660-663.

Ulyshen, M. D., Hanula, J. L., Horn, S., Kilgo, J. C. & Moorman, C. E. 2004. Spatial and temporal patterns of beetles associated with coarse woody debris in managed bottomland hardwood forests. *Forest Ecology and Management*, 199, 259-272.

United States Census Bureau. 2000. State and county Quickfacts. <http://quickfacts.census.gov>

Vander Wall, S. B., Kuhn, K. M. & Beck, M. J. 2005. Seed removal, seed predation, and secondary dispersal. *Ecology*, 86, 801-806.

Vanderklift, M. A. & Ponsard, S. 2003. Sources of variation in consumer-diet delta N-15 enrichment: a meta-analysis. *Oecologia*, 136, 169-182.

Vanhorne, B. 1983. Density as a misleading indicator of habitat quality. *Journal of Wildlife Management*, 47, 893-901.

Vitousek, P. M., Dantonio, C. M., Loope, L. L. & Westbrooks, R. 1996. Biological invasions as global environmental change. *American Scientist*, 84, 468-478.

Waddington, K. I., Bellchambers, L. M., Vanderklift, M. A., Walker, D. I. 2008. Western rock lobsters (*Panulirus cygnus*) in Western Australian deep coastal ecosystems (35–60 m) are more carnivorous than those in shallow ecosystems. *Estuarine, Costal and Shelf Science*, 79, 114-120.

Walton, B. M., Tsatiris, D. & Rivera-Sostre, M. 2006. Salamanders in forest-floor food webs: Invertebrate species composition influences top-down effects. *Pedobiologia*, 50, 313-321.

Wang, G. M., Wang, Z. W., Zhou, Q. Q. & Zhong, W. Q. 1999. Relationship between species richness of small mammals and primary productivity of arid and semi-arid grasslands in north China. *Journal of Arid Environments*, 43, 467-475.

Wang, G. M., Wolff, J., Vessey, S., Slade, N., Witham, J., Merritt, J., Hunter, M. & Elias, S. 2009. Comparative population dynamics of *Peromyscus leucopus* in North America: influences of climate, food, and density dependence. *Population Ecology*, 51, 133-142.

Wells-Gosling, N. & Heaney, L. R. 1984. *Glaucomys sabrinus*. *Mammalian Species*, 1-8.

White, G. C. & Burnham, K. P. 1999. Program MARK: survival estimation from populations of marked animals. *Bird Study*, 46, 120-139.

Whitford, W. G. 1976. Temporal fluctuations in density and diversity of desert rodent populations. *Journal of Mammalogy*, 57, 351-369.

Whittaker, J. C. & Feldhamer, G. A. 2005. Population dynamics and activity of southern short-tailed shrews (*Blarina carolinensis*) in southern Illinois. *Journal of Mammalogy*, 86, 294-301.

Wiener, J. G. & Smith, M. H. 1972. Relative efficiencies of 4 small mammal traps. *Journal of Mammalogy*, 53, 868.

Wiewel, A. S., Clark, W. R. & Sovada, M. A. 2007. Assessing small mammal abundance with track-tube indices and mark-recapture population estimates. *Journal of Mammalogy*, 88, 250-260.

Wilson, B. A., Robertson, D., Moloney, D. J., Newell, G. R. & Laidlaw, W. S. 1990. Factors affecting small mammal distribution and abundance in the eastern Otway Ranges, Victoria. *Proceedings of the Ecological Society of Australia*, 16, 379-396.

Wolff, J. O., Dueser, R. D. & Berry, K. S. 1985. Food-habits of sympatric *Peromyscus-leucopus* and *Peromyscus-maniculatus*. *Journal of Mammalogy*, 66, 795-798.

Wrazen, J. A. & Svendsen, G. E. 1978. Feeding ecology of a population of eastern chipmunks (*Tamias striatus*) in southeast Ohio. *American Midland Naturalist*, 100, 190-201.

Wu, H. Y. 1999. Is there current competition between sympatric Siberian weasels (*Mustela sibirica*) and ferret badgers (*Melogale moschata*) in a subtropical forest ecosystem of Taiwan? *Zoological Studies*, 38, 443-451.

Yahner, R. H. 1978. Burrow system and home range use by eastern chipmunks, *Tamias striatus* - ecological and behavioral considerations. *Journal of Mammalogy*, 59, 324-329.

Yarnell, R. W., Scott, D. M., Chimimba, C. T. & Metcalfe, D. J. 2007. Untangling the roles of fire, grazing and rainfall on small mammal communities in grassland ecosystems. *Oecologia*, 154, 387-402.

Yerger, R. W. 1955. Life history notes on the Eastern Chipmunk, *Tamias striatus lysteri* (Richardson), in central New York. *American Midland Naturalist*, 53, 312-323.

Yunger, J. A., Meserve, P. L. & Gutierrez, J. R. 2002. Small-mammal foraging behavior: Mechanisms for coexistence and implication for population dynamics. *Ecological Monographs*, 72, 561-577.

Zollner, P. A. & Crane, K. J. 2003. Influence of canopy closure and shrub coverage on travel along coarse woody debris by eastern chipmunks (*Tamias striatus*). *American Midland Naturalist*, 150, 151-157.

Zwolak, R. & Foresman, K. R. 2007. Effects of a stand-replacing fire on small-mammal communities in montane forest. *Canadian Journal of Zoology-Revue Canadienne De Zoologie*, 85, 815-822.

Appendix A. Vegetative cover (%) across plots; vegetative cover was estimated for each plot in 2009 by summing together percent coverage for each species across plot quadrats (each plot was divided into ten 1 x 1 meter quadrats). Treatment type is presented beneath plot labels; all A, B, and C-plots represent A, B, and C-slope positions. Data was contributed to this thesis by the Black Rock Forest Consortium**.**

		a1	a2	a3	a4	b1	b2	b3	b4	c1	c2	c3	c4
Species Name	**Common Name**	**N**	**O50**	**C**	**O**	**O50**	**O**	**N**	**C**	**O**	**C**	**O50**	**N**
Acalypha rhomboidea	Rhomboid Mercury	0	0	0	0	0	1	0	0	0	0	0	0
Acer pensylvanicum	Striped Maple	0	0	0	0	0	2	0.1	0	1	0	0	0.1
Acer rubrum	Red Maple	5	13.5	0.4	6.7	1.1	4.7	0.1	5.1	3.7	1	0.5	2.4
Acer saccharum	Sugar Maple	2.1	6.2	0.1	0	0	0	0.1	0	0.3	0	0	0
Ailanthus altissima	Tree Of Heaven	0	0.1	0	0	0	2.1	0	0	4.2	0	1	0
Ampelopsis brevipedunculata	Porcelain Berry	0	0	0	0	0.1	0	0	0	0	0	0	0
Arisaema triphyllum	Wild Turnip	3	6	0	0	0.2	0	0	0	0	0	0	0
Aster divaricatus	White Wood Aster	7	14.1	0.1	0.1	2.2	4.1	0	0	6.2	0.2	0.1	0
Avena sativa	Common Oat	0	0	0	1	0.1	0.1	0	0	0.3	0.1	0.3	0
Berberis thunbergii	Red Barberry	44	0	0	0	0	1.2	0	0	0	0	0	0
Betula allegheniensis	Yellow Birch	0	0	0	0	0.1	0	0	0	0	0	0	0
Betula lenta	Black Birch	2.1	5.6	2.4	54.1	0.5	2.4	3.6	1.4	11.3	24.7	2	2.9
Big Carex	Sedge	0	0	1	0	0	0	0	0	0	0	0	0
Bryophyta	Moss	55.2	153.8	115.2	270.3	11.7	16.6	111.41	61.2	6.7	14.1	51.9	57.1
Cardamine pratensis	Cuckoo Flower	14.1	32.3	0	41.1	0	1.2	0	0	1.2	0	0	0
Carex (other)	Sedge(Other)	60	22	0.2	3.2	0	4.3	1	0	3.3	0.1	0.1	0
Carex digitalis	Slender Wood Sedge	0	0	0	1	0	0	0	0	0	0	0	0
Carex pensylvanica	Pennslyvania Sedge	10	21	0.1	2	1.2	1	2	0	0	1.2	1	0.1
Carex platyphylla	Silver Sedge	1	2	0	3	0	1	0	0	4.1	0	0	0
Carex swannii	Swan's Sedge	0	1	0	0	0	0	0	0	0	0	0	0
Carya sp	Hickory Sp.	0.1	0.2	0	0	0	0.2	0	0	0.1	0	0.2	0
Celastrus orbiculatus	Oriental Bittersweet	1	2	0	0	0	1	1	0	0	0.1	0	0
Dennstaedtia punctilobula	Hay-Scented Fern	0	0	5	0.1	0	0	0	0	0.1	0	0	0
Dryopteris sp	Wood Ferns	0	0	0	1	0	0	0	0	0	0	0	0
Epilobium coloratum	Cinnamon Willow Herb	0	0	0	6	0	0	0	0	0	0	0	0

Equisetum arvense	Common Horsetail	0	0	0	8	0	0	0	0	0	0	0	0
Erechtites hieracifolia	American Burnweed	0	0	0	2.1	0	0	0	0	1	0	5	0
Euonymus alatus	Burningbush	5	10	0	0	0	0.1	0	0	0	0	0	0
Fagus grandifolia	American Beech	0	0	0.1	0	0	0	0	0	0	0	0	0
Fraxinus Americana	White Ash	15.1	31.3	0.2	0	14.8	3.1	0	0	2.1	0	0	0
Galium triflorum	Fragrant Bedstraw	7.2	14.4	0	1	2	0	0	0	0	0	0	0
Gaylussacia baccata	Black Huckleberry	0	0	5	47.1	0	0	46.1	48.1	0	0	0	117
Hamamelis virginiana	Witchhazel	2.2	5.7	1.4	3	26.2	2.3	2	2	4.1	12.7	2.1	1.3
Juncus sp	Rushes Sp.	0	1	1	0	0.1	0	0	0	0	0	0	0
Kalmia latifolia	Mountain-Laurel	0	0	0	0	0	0	1	0	0	0	0	0
Leersia virginica	Whitegrass	23	46	0	0	0	0	0	0	0	0	0	0
Liriodendron tulipifera	Tulip Tree	4.1	10.3	1.1	7.4	0	1.6	0.2	0	11.9	0	0.2	0.4
Lobelia inflate	Indian Tobacco	0	0	0	1	0	0	0	0	0	0	0	0
Lonicera japonica	Japanese Honeysuckle	2	4	0	0	0	0	0	0	0	0	0	0
Maianthemum canadense	Canadian May-Lily	0	1	0	0	0	0	0	0	0	0.1	0	0
Medeola virginiana	Indian Cucumber-Root	0	0	0.1	0	0	0	0	0	0	0	0	0
Microstegium vimineum	Japanese Stiltgrass	90.2	225.4	0.1	6	0	1.3	0	0	0	0	0	0
Mitchella repens	Partridgeberry	8.1	16.2	2.2	0	2	1.2	0	0	2	0.1	0	0
Monotropa hypopithys	Yellow Bird's-Nest	0	0	0.1	0	0	0	0	0	0	0	0	0
Monotropa uniflora	Ghost Plant	2	4	0.1	0	0	0	0	1	0	0	0	0
Nyssa sylvatica	Black Tupelo	17	44	2	15.2	0	35.5	0	0	0	0	0	11.2
Osmundea cinnamomea	Cinnamon Fern	2	4	0	0	0	0	0	0	0	0	0	0
Oxalis sp	Wood-Sorrel Sp.	0	0	0	1	0	0	0	0	0	0	0	0
Parthenocissus quinquefolia	Virginia Creeper	7.2	15.6	0.1	0.4	0.7	2.1	0.1	0	0.2	0	0.2	0.1
Pilea pumila	Clearweed	2	4.1	1.1	61	0	0	0	0	0	0	0	0
Poaceae sp (other)	Grass	4.1	10.2	0.1	23	0.3	0.2	0	0	0	0	0	0
Polygonum lapathifolium	Curlytop Knotweed	0	0	0	1	0	0	0	0	0	0	0	0
Polygonum sagittatum	Arrowleaf Tearthumb	0	0	0.1	17	0	0	0	0	0	0	0	0
Polygonun punctatum	Dotted Knotweed	2	4	0	1	0.2	8	0	0	0	0	0	0
Polystichum acrostichoides	Christmas Fern	54	111	4.1	1	2.2	3.1	0	0	13.2	4	3.1	0
Potentilla sp	Typical Cinquefoils Sp.	1	2	0	1.1	0	0	0	1	0	0	0	0
Prenanthes alba	White Rattlesnake Root	3	6	0.1	0	0	0.1	0	1	0	0	0.1	0
Prunus serotina	Black Cherry	0	0	0	0	0	0	0	0	2	0	0	0
Quercus alba	White Oak	0	0	0	0	0	0	0.1	0.1	0	0	0	0
Quercus prinus	Chestnut Oak	0	0	0	0	0	0	1	8.5	0	0	0	0.1
Quercus rubra	Northern Red Oak	0	0	0	0	4	0.1	0	0	0	0	1	0

Rhamnus cathartica	Buckthorn	0	0	0	0	2	0	0	0	0	0	0	0
Rosa multiflora	Multiflora Rose	0	0	0	1	0	0	0	0	0	0	0	0
Sassafras albidum	White Sassafras	1.1	2.2	1.1	1	0	0	0	0	0	0	0	0
Solidago	Goldenrod	5	10.1	0	0	0	0	0	0	2	0	0	0
Streptopus amplexifolius	Clasping Twistedstalk	0	0	3.3	1	0.1	0	0	0	1.1	0.1	0	0.1
Thelypteris noveboracensis	New York Fern	27	75	12	178	0	0.1	0	0	0	0	0	0
Thelypteris palustris	Marsh Fern	9	18	1	0	4.1	0	0	0	0	0	6	0
Toxicodendron radicans	Poison Ivy	1.1	3.3	0	2	0.3	3.3	0	0	2.3	0	0.3	0
Trillium sp	Trillium Sp.	0	0	0	0	0	0	0	0	0.1	0	0	0
Tsuga Canadensis	Canadian Hemlock	0.1	0.2	0	2	1	0	0	0	0	0	0	0
Unknown	n/a	0	0	0	19	0.2	0	1	7.2	0.1	0.3	0.2	0
Uvularia sessilifolia	Sessileleaf Bellwort	0	1	0	0	0	0	0	0	0	0	0	0
Vaccinium angustifolium	Lowbush Blueberry	0	0	0	0	0.1	0	1	0	0	0	0.1	0
Vaccinium pallidum	Blue Ridge Blueberry	0	2	39.3	20.1	2	0	27.1	129.1	44.3	1	4.3	16.6
Veronica officionalis	Common Speedwell	5.1	10.2	0	0	0.1	0.3	0	0	0	0	0	0
Viburnum acerifolia	Mapleleaf Viburnum	0	2	0	0	0	0	1	1.1	1	0	0	0.1
Viburnum prunifolium	Blackhaw Viburnum	1	3	0	0	0	0	0	0	0	0	0	0
Viola sororia	Wolly Blue Violet	0	0	0.1	0	0.1	0	0	0	0	0	0	0
Vitis	Grapevines	3.1	6.2	0	0.2	0	2.1	0	0	1.2	0	0	0
Sum		453.1	829.4	85.1	540.9	68	90.8	88.5	205.6	124.4	45.7	27.8	152.4
Percent Coverage*		**22.7**	**41.5**	**4.3**	**27.0**	**3.4**	**4.5**	**4.4**	**10.3**	**6.2**	**2.3**	**1.4**	**7.6**
Mean Canopy Openness**		**2.61**	**2.84**	**2.66**	**1.69**	**2.66**	**2.49**	**2.51**	**2.51**	**1.97**	**4.21**	**2.47**	**2.40**

*Percent cover was calculated by dividing total plot coverage by 2000 and multiplying this number by 100.

** Canopy openness = the amount of light reaching the forest floor

Appendix B. Species Diversity (H) values across plot, treatment type, and slope position. Species diversity was calculated using the Shannon-Weiner diversity index.

Plot	Treatment	Slope Position	Species Diversity
a1	N	lower	0.85
a2	O50	lower	0.95
a3	C	lower	0.65
a4	O	lower	0.84
b1	O50	middle	0.65
b2	O	middle	0.60
b3	N	middle	0.51
b4	C	middle	0.66
c1	O	upper	0.70
c2	C	upper	0.40
c3	O50	upper	0.85
c4	N	upper	0.72

Appendix C. Prey items sampled at Black Rock Forest across various treatment types and slope positions. Superscripts indicate values taken from the literature to supplement data collected in the current experiment; where a = Bennet and Hobson (2009), b = Flaherty, Ben-David, and Smith (2010), c. = Koenig et al. (2008), and d = Trudell et al. (2004).

Prey Guild	**Taxonomic Classification**	**Common Name**	**n**
Berries	*Berberis vulgaris*	Barberry	4
	Vacinium	Blueberry	3
	Gaylussacia baccata	Huckleberry	10
	Rubus phoenicolasius	Wineberry	3
Invertebrates	Aranea[a]	Spider	6
	Coleoptera[a]	Beetle	53
	Diptera[a]	Fly	27
	Diplopoda[b]	Millepede	30
	Lumbricina[b]	Earthworm	17
	Formicidae	Ant	8
	Lepidoptera	Moth	69
Nuts	*Carya*	Hickory	5
	Quercus[c]	Acorn	17
Fungi[d]	Fungi	ECM Fungi	23