Small Mammal Species Richness And Relative Abundance Across The Elevational Gradient And Microhabitats Of Pine Mountain, Eastern Kentucky

Sarah Elizabeth Baker
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SMALL MAMMAL SPECIES RICHNESS AND RELATIVE ABUNDANCE ACROSS THE ELEVATIONAL GRADIENT AND MICROHABITATS OF PINE MOUNTAIN, EASTERN KENTUCKY

BY

SARAH ELIZABETH BAKER

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SMALL MAMMAL SPECIES RICHNESS AND RELATIVE ABUNDANCE
ACROSS THE ELEVATIONAL GRADIENT AND MICROHABITATS OF PINE
MOUNTAIN, EASTERN KENTUCKY

BY

SARAH ELIZABETH BAKER

Submitted to the Faculty of the Graduate School of
Eastern Kentucky University
in partial fulfillment of the requirements for the degree of

MASTER OF SCIENCE

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ABSTRACT

Identifying microhabitat conditions influencing species abundance is critical to understanding distributions of species. Pine Mountain has forest types uncommon in Kentucky, but systematic small mammal surveys are dated. My objectives were: investigate variation in small mammal occurrence on Pine Mountain, and update small mammal records in the Eastern Kentucky University (EKU) mammal collection. I deployed pitfall traps, open from May - November 2021, at four state nature preserves (SNPs) on Pine Mountain. I measured microhabitat using 1 m² quadrats at traps, then used redundancy analysis and generalized linear mixed (GLM) models to determine which conditions best explained variation in occurrence of small mammals. Most of the 95 small mammals I captured were at Bad Branch SNP and Blanton Forest SNP (79%).

From the redundancy analysis, canopy openness, number of *Kalmia latifolia* stems, number of rocks, leaf litter depth, and elevation explained the most variation in abundances of small mammals. *Sorex fumeus* was more abundant at microsites with deeper leaf litter and more rocks, and less at lower elevation microsites with less canopy openness. *Microtus pinetorum* abundance was greater at microsites with greater canopy openness, more *K. latifolia* stems, and less rocky debris. GLM models suggest leaf litter depth, number of rocks, and soil temperature best explained variation in small mammal richness, whereas variation in total abundance was best explained by rocky debris, *Rhododendron* cover, and soil temperature. Historically, 13 records in the EKU small mammal collection originally came from Harlan and Letcher counties, out of 442 records collected in the state. Novel specimens I collected and accessioned into the EKU small mammal collection included *S. fumeus* and *Sorex hoyi* from Harlan and Letcher counties.
This investigation updates EKU’s museum efforts, as well as small mammal records on Pine Mountain.
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Chapter 1: Small mammal richness and relative abundance across the elevational gradient and microhabitats of Pine Mountain in Eastern Kentucky

Introduction

Associations of fauna with biotic and abiotic factors, such as elevation, vegetation, and microhabitat characteristics interest landscape ecologists because of the factors’ direct link with global distributions of species. For example, montane systems have elevational gradients and, often, mosaics of forest condition (Braun 1935, 1942, Van Miegroet et al. 2005). The variation of abiotic factors along elevational gradients allows for elevational studies to be conducted within local montane systems and applied broadly (Stevens et al. 2019). Further, variation in microhabitat characteristics of montane systems across elevation (Braun 1935) results in habitat heterogeneity (Gómez et al. 2004). Habitat heterogeneity is also related to species richness, though the relationship differs among faunal groups (Tews et al. 2004). Therefore, montane systems are useful for investigating the distribution and abundance of fauna.

Precipitation, temperature, and exposure to solar radiation vary with elevation and slope aspect in montane systems. Specifically, volume and distribution of precipitation is dependent on mountain size and shape (Houze 2012). On larger mountains, the volume of precipitation is negatively correlated with elevation because air temperature decreases with increasing elevation (Houze 2012). As a complication, precipitation generally increases with increasing elevation on temperate mountains reaching over 3500 m above sea level (ASL) (Rowe 2009). Most clouds precipitate out before reaching downwind slopes and solar radiation on downwind slopes can cause cloud evaporation that contributes to drier conditions than on upwind slopes (Houze 2012). Additionally, the
intensity of solar radiation varies with slope aspect, with southern-facing slopes generally exposed to higher solar radiation (Braun 1935). For example, Pine Mountain, a ridge within the Cumberland Mountains in southeastern Kentucky (Braun 1935), ranges in elevation from about 370 m to about 850 m ASL. On Pine Mountain, southeastern slopes receive higher solar radiation and are sheltered from winds, resulting in drier conditions than on northwestern slopes (Braun 1935).

Over the past 400 years, forest composition in the eastern United States has shifted from predominantly xeric to mesic forests because of a widespread process termed ‘mesophication’ (Nowacki and Abrams 2008). Mesophication is due in part to fire suppression within historically fire-sensitive forests dominated by pyrophytes (Nowacki and Abrams 2008). Resulting mesic forest canopies tend to have greater closure than xeric forests (McShea et al. 2003, Nowacki and Abrams 2008), facilitating further mesophication by inhibiting establishment of shade-intolerant pyrophytes (Nowacki and Abrams 2008). This shift in forest types is occurring in southeastern Kentucky (Nowacki and Abrams 2008).

The leaf litter associated with forest types can influence moisture levels. Mesic forests have more fire-resistant trees in the canopy, more absorbent leaf litter, and more tree species than xeric forests (Kreye et al. 2013). Leaf litter in mesophytic forests is fire-resistant because it absorbs and retains more moisture than pyrophytic leaf litter (Kreye et al. 2013). In contrast, xeric forests tend to be dominated by fire-adapted trees in the canopy and have drier leaf litter (Kreye et al. 2013). This partly contributes to the positive feedback system driving mesophication. The differences in vegetative assemblage between mesic and xeric forests (McShea et al. 2003, Nowacki and Abrams
2008), therefore, can help to maintain moisture levels that contribute to mesophication within forests.

The Appalachian Mountains extend from Georgia to Maine, with a broad elevational range and varied degree of forest mosaicism (Braun 1935). Pine Mountain has many distinct forest types that differ in their composition of dominant canopy and understory species (Quinlan et al. 2007, 2009, Napier 2020); some of these forest types, including pine barren forests, are uncommon in Kentucky and the Appalachian Mountain Range (Quinlan et al. 2009). Given the general north-south alignment of the ridgeline, forest type on Pine Mountain varies with slope aspect (Braun 1935). Specifically, forests on the northwestern-facing slopes are generally more mesic, whereas forests on the southeastern-facing slopes are generally more xeric (Braun 1935).

Variable elevations and forest types within montane ecosystems in North America allow for investigations of small mammal species diversity and abundance across a heterogeneous landscape (Brannon 2002, Rowe 2009). For example, Ford et al. (2005) found that species richness of shrews within the Appalachian Mountains increased between 160 and 1600 m ASL. In Utah, Rowe (2009) surveyed small mammal species richness across an elevational gradient of 1500 to 3500 m ASL, and found that richness peaked at about 2500 m ASL. Small mammals have been used to investigate fine-scale variation in species richness and abundance because elevational range (Ford et al. 2005), and habitat preference (Barbour and Davis 1974) vary among species.

Small mammal diversity on Pine Mountain is relatively high, with 15 small mammal species historically documented (Error! Reference source not found.). Based on records reported by Ford et al. (2005) in other portions of Appalachia, some species
documented on Pine Mountain could be considered habitat generalists, found broadly across elevations; others are restricted to narrower elevational ranges. For example, rock shrews (*Sorex dispar*) and water shrews (*Sorex palustris*) have been recorded to occur from 610 to 1538 m ASL, southern short-tailed shrews (*Blarina carolinensis*) from 160 to 553 m ASL, and northern short-tailed shrews (*Blarina brevicauda*) and pygmy shrews (*Sorex hoyi*) from 160 to 1600 m ASL (Ford et al. 2005). Elsewhere in the southern United States, deer mice (*Peromyscus maniculatus*) and southern red-backed voles (*Myodes gapperi*) have been trapped at high elevations (>1000 m ASL) (Rickart 2001, Trani et al. 2007). Southern bog lemmings (*Synaptomys cooperi*), although most strongly associated with wetlands, were documented to inhabit mid to high elevation habitat (Trani et al. 2007).

Small mammal species presence and distribution have also been reported to be influenced by forest type (Trani et al. 2007). Some species, such as masked shrews (*Sorex cinereus*), are habitat generalists with respect to forest type, while other species are more abundant in xeric forests (Barbour and Davis 1974, Trani et al. 2007). In the central and southern Appalachian Mountains, Ford et al. (2005) reported capturing masked shrews and smoky shrews (*Sorex fumeus*) most frequently in xeric forests. McShea et al. (2003), however, did not report trapping either species more frequently in xeric or mesic forests in the southern Appalachian Mountains of Virginia. Ford et al. (2005) also trapped northern short-tailed shrews and pygmy shrews with equal frequency in both mesic and xeric forest types.

Discrepancies concerning the forest types preferred by small mammal species may be explained by the influences of microhabitat characteristics, such as vegetative
cover and density of coarse woody debris (CWD) (Trani et al. 2007) (Table 2). Some species, such as southern bog lemmings and water shrews, are more commonly associated with specific habitat features rather than forest types (Trani et al. 2007). The presence of small mammals is reported to be positively correlated with presence of CWD (McCay et al. 1998, Greenberg 2002). The cover provided by CWD and the roots of fallen trees have been reported to be important habitat for certain small mammal groups, such as shrews (Greenberg 2002). Therefore, small mammal species richness and abundance may reflect both forest type and microhabitat features.

This investigation aimed to survey small mammal communities at four nature preserves on Pine Mountain. My objectives were to assess: 1) variation in species richness and abundance of small mammals across elevational gradients on Pine Mountain, and 2) differences in species richness and abundance of small mammals with respect microhabitat conditions found on Pine Mountain. By surveying small mammal communities across an elevational gradient and among forest types, I assessed the extent to which these factors and associated habitat variables influenced small mammal abundance within the unique topography of Pine Mountain.

**Methods**

*Site Description*

The Cumberland Mountains are located along Kentucky’s border with Virginia. Pine Mountain lies within the Level III Central Appalachian and Level IV Cumberland Mountain Thrust Block ecoregions (Woods et al. 2002). Pine Mountain stretches over 200 km from Whitley County to Pike County (McGrain 1983, Kleber and Kentucky Bicentennial Commission 1992) along the western portion of the Cumberland Mountains.
Coal deposits and soils, developed from oceanic deposits during the Pennsylvanian and Mississippian Periods, characterize the geology of the Cumberland Mountains (McGrain 1983). On Pine Mountain, the soils have high sand content (Braun 1935). Soil composition and vegetative communities vary at the microscale along the length of Pine Mountain (Braun 1935), contributing to the diversity of forests documented on Pine Mountain.

The Cumberland Mountains contain multiple, distinct forest types as a result of topography and climate. Rock formations in the region are primarily sandstone (Woods et al. 2002), such as the Lee and Pennington formations (Kleber and Kentucky Bicentennial Commission 1992). Average winter temperatures range from -6.7°C to 6.7°C, and average summer temperatures range from 16.7°C to 30°C (Woods et al. 2002). Forests in the Cumberland Mountains are primarily wet and humid; and average annual precipitation can exceed 140 cm (Woods et al. 2002). Mixed-mesophytic and mesic forests are common and contain a diversity of deciduous species (Woods et al. 2002). However, although mesic forests types are common, xeric forests are still important forest types (Braun 1935). The topography and climate surrounding Pine Mountain creates an interesting mosaic of forests.

Pine Mountain contains forest types and stand conditions uncommon in other parts of the Cumberland Mountains (Braun 1935). Southeast-facing slopes have primarily xeric forests with canopies dominated by pitch pine (Pinus rigida), yellow pine (Pinus echinata), scrub pine (Pinus virginiana), chestnut oak (Quercus montana), and tulip poplar (Liriodendron tulipifera) (Braun 1935). Historically, American chestnut (Castanea dentata) was also a dominant canopy species (Braun 1935), but is now
functionally extinct in this region. The understories on southeast-facing slopes are
dominated by shrubs, including mountain laurel (*Kalmia latifolia*) and deerberry
(*Vaccinium stamineum*) (Braun 1935). In contrast, northwest-facing slopes on Pine
Mountain primarily support mesophytic vegetation (Braun 1935). Mesic forest canopies
are dominated by tulip poplar, white oak (*Quercus alba*), cucumber magnolia (*Magnolia
acuminata*), black locust (*Robinia pseudo-acacia*), sugar maple (*Acer saccharum*),
basswood (*Tilia heterophylla*), and common buckeye (*Aesculus octandra*) (Braun 1935).
The understories are dominated by shrubs, such as rhododendron (*Rhododendron* spp.),
and ferns, such as cinnamon fern (*Osmunda cinnamomea*) (Braun 1935).

Xeric forests on Pine Mountain are most common on the mountain’s southeast-
facing slopes and are only observed on the northwest-facing slope at the High Rock
summit, where thinner soils are present (Braun 1935). Vegetation historically
documented on High Rock summit, such as purple-stem cliffbrake (*Pellaea
atropurpurea*) and bottlebrush grass (*Hystrix patula*), are adapted to dry conditions and
thinner soils (Braun 1935). Excluding the summit at High Rock, all forests located on the
northwest-facing slope are mesic (Braun 1935). Mesic forest can also be found on
southeast-facing slopes where small valleys with deeper soils support diverse, deciduous
tree species (Braun 1935).

I conducted my investigation at four state nature preserves (hereafter, SNPs)
located in southeastern Kentucky on Pine Mountain: Blanton Forest, Bad Branch, Hi
Lewis, and Kingdom Come SNPs. Bad Branch SNP and Kingdom Come SNP are located
in Letcher County, Blanton Forest SNP in Harlan County, and Hi Lewis SNP in Harlan
and Letcher Counties (Figure 1). All field sites used in this investigation are under the
The stewardship of the Office of Kentucky Nature Preserves (OKNP). The OKNP protects these natural areas because of their cultural and scientific value (Kleber and Kentucky Bicentennial Commission 1992).

The approximately 1250 ha Blanton Forest SNP, located on the south-facing side of Pine Mountain, contains the largest amount of diverse and old-growth forest found in Kentucky (Quinlan et al. 2009). A survey of the site revealed it was composed of mesic and xeric forests. The mesic forests were characterized by eastern hemlock (*Tsuga canadensis*) and tulip poplar in the overstory and rhododendron in the understory (Quinlan et al. 2009). The xeric forests were characterized by oaks (*Quercus* spp.) in the overstory and mountain laurel and mixed brambles (e.g., *Rubus* spp. and *Smilax* spp.) in the understory (Quinlan et al. 2009). White oak, basswood, and sugar maple were also common canopy species in mesic forest canopies, while pitch pine, shortleaf pine (*Pinus echinata*), and Virginia pine (*Pinus virginiana*) were common in xeric forest canopies (Quinlan et al. 2009).

The approximately 290 ha Hi Lewis SNP study site contains pine-oak and pine barren forests (Napier 2020). Primarily xeric, pine barren forest, Hi Lewis SNP contains pine savannah woodlands, sub-xeric forests, and Appalachian pine-oak forests (Napier 2020). The forests were dominated by pitch pine, shortleaf pine, chestnut oak, and red maple (*Acer rubrum*) (Napier 2020). The understory was dominated by little bluestem (*Schizachyrium scoparium*) and bracken fern (*Pteridium aquilinum*) (Napier 2020). Species including mountain laurel and sassafras (*Sassafras albidum*) were common in the midstory (Napier 2020). Hi Lewis SNP also contains state endangered vegetation, such as
yellow wild indigo (*Baptisia tinctoria*) (Napier 2020). Hi Lewis SNP provided access to the top of Pine Mountain via the Little Shepherd Trail (Highway 1679).

Kingdom Come SNP is east of all field sites except Bad Branch SNP and provided access to the north-facing slope of Pine Mountain (Kentucky Energy and Environment Cabinet n.d.). Most notably, Kingdom Come SNP includes a north-facing slope that allowed for study site replication across north- and south-facing slopes. Although Kingdom Come SNP was not the only site that provided north- and south-facing slopes, Kingdom Come SNP had the only accessible north-facing slope. Kingdom Come SNP is accessible from the top of Pine Mountain via the Little Shepherd Trail (Highway 1679).

The approximately 1070 ha Bad Branch SNP field site is predominantly on the south-facing slope of Pine Mountain (Quinlan et al. 2007). A survey of the site revealed that it was composed of mesic forest characterized by a rhododendron dominated understory with a mixed hardwood canopy, whereas xeric forest was characterized by a mountain laurel dominated understory and oak-pine dominated canopy. The canopies of mesophytic forests were dominated by hemlock, tulip poplar, and yellow birch (*Betula allegheniensis*) (Quinlan et al. 2007). The understories of these forests were dominated by rhododendron and hemlock (Quinlan et al. 2007). Xeric forests were dominated by red maple, tulip poplar, chestnut oak, pitch pine, shortleaf pine, and Virginia pine (Quinlan et al. 2007). Rhododendron and mountain laurel were common understory species (Quinlan et al. 2007). Bad Branch SNP is accessible from the base of the field site, just off Highway 932 (Quinlan et al. 2007).
Trapping Design

I used pitfall traps to collect reclusive shrews and other small mammals within 20 m × 20 m plots stratified across all study sites. Plots were separated by at least 15 m within sites to ensure small mammal populations were statistically independent (Findley and Yates 1991). Replicate plots were placed at low (≤550 m), medium (>550 m and ≤700 m), and high (>700 m) elevation areas, based on accessibility, at each field site (Table 3). I established mid and low elevation plots at Blanton Forest SNP ranging from 469 m to 616 m ASL, mid and high elevation plots at Hi Lewis SNP ranging from 617 m to 757 m ASL, high elevation plots at Kingdom Come SNP ranging from 789 m to 790 m ASL, and mid and low elevation plots at Bad Branch SNP ranging from 538 m to 641 m ASL (Table 3). I determined the elevation of each plot using a Garmin 64s GPS unit.

Trap success is typically low when trying to capture small mammal species (Mengak and Guynn 1987, McCay et al. 1998, Brannon 2002). Trap type and location are therefore important when considering sample size and statistical analysis of data. Pitfall traps catch greater numbers and diversity of small mammals than do conventional small mammal traps (Williams and Braun 1983). Adding drift fences to pitfall trap arrays may increase trap success for certain small mammal groups, such as shrews (Williams and Braun 1983). Using natural features (e.g., logs, rocks, and tree stumps) as drift fences has been reported to increase trap success (McCay et al. 1998). Within plots, I placed 10 pitfall traps [trap = 2 L plastic container (17 cm depth × 15 cm diameter)] next to natural drift fences (}
Figure 2). Pitfall trapping was conducted May through November of 2021; traps were checked bi-weekly in May and June, and then monthly thereafter. Holes were drilled around the rim of pitfall traps to enhance drainage and reduce flooding from rainfall. In addition, plastic plates were used to cover pitfall traps and prevent flooding (Figure 3). Debris was used to prop up the covers and allow small mammals to enter the traps. Pitfall traps were filled with 500 ml of 95% ethanol mixed with 500 ml of water, and enough mineral oil to cover the surface. Mineral oil was added to reduce evaporative loss and accelerate drowning of captured animals (Howard and Brock 1961, Sikes and Gannon 2011). This approach is consistent with ongoing pitfall trapping efforts currently used by the Kentucky Department of Fish and Wildlife Resources (KDFWR) as part of their small mammal survey efforts in the Cumberland Mountains (pers. comm., Zack Couch, Nongame Program Coordinator, KDFWR). Furthermore, these methods met with documented approval from the KDFWR, the OKNP, and Eastern Kentucky University’s (EKU) Institutional Animal Care and Use Committee (Protocol 02-2021).
**Habitat Assessment**

Within each 20 m x 20 m plot, I measured microhabitat characteristics within a 1 m$^2$ quadrat with one side placed adjacent to the drift fence object.
Habitat measurements were taken once per month and when traps were checked at each trap location. Within the 1 m² quadrat, the following parameters were assessed once per month: soil temperature, soil saturation, canopy closure, number of rhododendron and mountain laurel stems rooted in the plot, the percent rhododendron and mountain laurel cover, leaf litter depth, and duff depth. Total length, maximum width, and maximum height of all CWD and rocky debris within each 1 m² quadrat (Davis et al. 2010) were assessed at each trap location during the months of May, June, and July. I visually estimated understory cover and counted the number of rhododendron and mountain laurel plants rooted within quadrats (Fritts et al. 2015). At the center of each quadrat, I used a spherical densiometer to assess canopy closure (Greenberg et al. 2007). At each corner of the quadrat, I measured leaf litter and duff depth (Greenberg et al. 2007), soil temperature with a soil thermometer, and percent soil moisture with a Kelway soil pH and moisture meter (Kel Instruments Company).
Statistical Analyses

I used the coding program R (2019) for all statistical analysis and data visualization. For these analyses, I used R packages stats (R Core Team 2019), ggplot2 (Wickham 2016), sf (Pebesma 2018), dplyr version 1.0.3 (Wickham et al. 2021), ggmap version 5 (Kahle and Wickham 2013), ggsn version 0.5.0 (Baquero 2019), raster version 3.4-13 (Hijmans 2021), BiodiversityR (Kindt and Coe 2005), car (Fox and Weisberg 2019), lme4 (Bates et al. 2015 p. 4), MuMIn version 1.46.0 (Bartoń 2022), and rworldmap (South 2011). A full transcript of all code and output were provided to the principal sponsor of this work (OKNP). Microhabitat measurements were averaged across the months of data collection for each trap because seasonality was not a variable of interest. Body standard measurements were also averaged by species for each small mammal species collected.

To determine how small mammal community composition differed with microhabitat conditions at trap locations, I used a redundancy analysis. To determine the predictor variables that best explained variation in the data, I constructed a full model containing all microhabitat conditions as predictor variables. Relative abundance of small mammal species, calculated using the Hellinger transformation on the species abundance data for each trap, was used as the response variable. The Hellinger transformation is recommended when conducting a redundancy analysis on community data, because it linearizes the data without compromising resolution (Legendre and Gallagher 2001). Stepwise model selection was used to identify statistically significant microhabitat variables for inclusion in the constrained ordination. The full model used at the initiation of stepwise was a model that contained all microhabitat conditions.
In order to understand how small mammal species richness and total abundance were influenced by microhabitat conditions at trap location, I used an information theoretic approach. Candidate models were generalized linear mixed models with a Poisson error distribution, and field site as the random effect. For each response variable, I created a candidate model set that included all possible combinations of microhabitat conditions as predictor variables. I removed variables with variance inflation factors greater than five. I used lower AICc values to indicate the model best fit to the data, and models with ΔAICc less than two were considered competing (Burnham and Anderson 1998).

Results

Summary of Microhabitat Variables

Rocky debris length, width, and height differed among the state nature preserves (Figure 4). Fallen boulders were primarily responsible for the observed outliers for the height of rocky debris within quadrats. The average length of rocky debris was 37.0 ± 213.5 cm (mean ± std. dev.) at Bad Branch SNP, 35.6 ± 45.1 cm at Blanton Forest SNP, 32.2 ± 62.1 cm at Hi Lewis SNP, and 56.5 ± 47.8 cm at Kingdom Come SNP. Width of rocky debris, however, had greater variation among the state nature preserves. Kingdom Come SNP had a higher average width (25.5 ± 14.6 cm) of rocky debris than Bad Branch SNP (19.7 ± 45.0 cm), Blanton Forest SNP (19.1 ± 18.1 cm), and Hi Lewis SNP (15.2 ± 22.0 cm). The greatest amount of within and between site variation was observed for height of rocky debris measured within quadrats. Kingdom Come SNP had a higher average height (17.5 ± 7.3 cm) of rocky debris than Bad Branch SNP (9.1 ± 11.5 cm), Blanton Forest SNP (14.7 ± 22.4 cm), and Hi Lewis SNP (7.6 ± 12.5 cm). The height of
rocky debris at Blanton Forest SNP was also higher than that at Hi Lewis SNP and Kingdom Come SNP.

There was little variation in CWD measurements across sites (Figure 4). CWD included debris as large as fallen trees and as small as fragments of branches and bark, accounting for the outliers observed. The average length of CWD was 48.9 ± 80.5 cm at Bad Branch SNP, 39.4 ± 88.3 cm at Blanton Forest SNP, 46.3 ± 132.5 cm at Hi Lewis SNP, and 51.1 ± 57.3 cm at Kingdom Come SNP. Similarly, average width of CWD did not vary among sites. Average width of CWD was 5.2 ± 8.8 cm at Bad Branch SNP, 4.7 ± 8.9 cm at Blanton Forest SNP, 5.2 ± 13.9 cm at Hi Lewis SNP, and 2.7 ± 23.1 cm at Kingdom Come SNP. Height of CWD was relatively low across sites, with an average of 2.8 ± 4.6 cm at Bad Branch SNP, 2.1 ± 4.2 cm at Blanton Forest SNP, 2.3 ± 4.8 cm at Hi Lewis SNP, and 1.8 ± 3.2 cm at Kingdom Come SNP.

Sites varied in the amount of canopy and understory cover (Figure 5). Canopy openness was generally low across study areas, but was greater at Hi Lewis SNP (4.9 ± 6.0 %) than at Bad Branch SNP (1.5 ± 1.8 %), Blanton Forest SNP (1.5 ± 2.0 %), and Kingdom Come SNP (1.2 ± 1.6 %). Regarding understory cover data (Figure 6), plots at Kingdom Come SNP (63.3 ± 42.1 %) had greater average rhododendron cover than plots at Bad Branch SNP (29.4 ± 38.4 %) and Hi Lewis SNP (0.7 ± 3.5 %). Blanton Forest SNP (44.6 ± 42.2 %) also had greater average rhododendron cover than Hi Lewis SNP. Save occasional outliers at Hi Lewis SNP and Kingdom Come SNP, there was no variation in percent mountain laurel cover among sites, and cover was low overall (Figure 6). The average mountain laurel cover was 0.0 ± 0.1 % at Blanton Forest SNP, 1.2 ± 5.5 % at Hi Lewis SNP, and 6.1 ± 14.4 % at Kingdom Come SNP. There was no mountain
laurel within plots at Bad Branch SNP. Additionally, save occasional outliers at Hi Lewis SNP, there was no variation in the number of rhododendron or mountain laurel stems rooted within quadrats among sites (Figure 6). The average number of rhododendron stems rooted in quadrats was 0.2 ± 0.6 at Bad Branch SNP, 0.3 ± 0.7 at Blanton Forest SNP, 0.0 ± 0.1 at Hi Lewis SNP, and 0.4 ± 0.9 at Kingdom Come SNP. The average number of mountain laurel stems rooted in quadrats was 0.1 ± 0.3 at Hi Lewis SNP. There were no mountain laurel stems rooted within quadrats at Bad Branch SNP, Blanton Forest SNP, or Kingdom Come SNP (Figure 6).

Soil conditions differed between the state nature preserves (Figure 7). Soil duff was deeper, on average, at Kingdom Come SNP (12.8 ± 4.1 cm) than at Bad Branch SNP (5.1 ± 3.5 cm), Blanton Forest SNP (8.4 ± 5.1 cm), and Hi Lewis SNP (5.5 ± 4.2 cm). Duff was also deeper at Blanton Forest SNP than Bad Branch SNP and Hi Lewis SNP. In addition, Kingdom Come SNP (5.6 ± 2.4 cm) had deeper leaf litter than Bad Branch SNP (3.9 ± 3.0 cm), Blanton Forest SNP (3.9 ± 2.9 cm), and Hi Lewis SNP (3.7 ± 3.2 cm). Leaf litter depth did not vary among the other sites. Blanton Forest SNP had higher average soil saturation (48.7 ± 18.8 %) than Bad Branch SNP (40.0 ± 14.4 %), Hi Lewis SNP (36.9 ± 16.6 %), and Kingdom Come SNP (32.5 ± 15.0 %). Soil temperature varied little among sites. Average soil temperature was 13.9 ± 3.5 ºC at Bad Branch SNP, 14.7 ± 4.2 ºC at Blanton Forest SNP, 14.3 ± 4.5 ºC at Hi Lewis SNP, and 16.8 ± 2.1 ºC at Kingdom Come SNP.

**Summary of Trapping Effort**

In total, survey efforts yielded 24,472 trap nights (TN; a single night over which a trap was deployed and operable) across all sites. I sampled a total of 7,550 TN at Bad
Branch SNP, 7,750 TN at Blanton Forest SNP, 7,381 TN at Hi Lewis SNP, and 1,791 TN at Kingdom Come SNP. A total of 94 traps were rendered inoperable at various points throughout the field season; of these, 10 at Bad Branch SNP, 15 at Blanton Forest SNP, 19 at Hi Lewis SNP, and 50 at Kingdom Come SNP. Most trap disturbances occurred in the spring months, May through June, and early fall months, September through October (Figure 8). I captured 95 small mammals, yielding an overall catch per unit effort of 0.4 per 100 TN. Catch per unit effort across sites were as follows: 0.6 for Bad Branch SNP, 0.4 for Blanton Forest SNP, 0.2 for Hi Lewis SNP, and 0.3 for Kingdom Come SNP.

In this study, small mammal captures across genera included 73 Sorex, 10 Peromyscus, six Microtus, four Blarina, and two Myodes (Figure 9). Further, a total of eight species were recorded across sites (Table 4). Across sites, Sorex accounted for the most captures collected at Bad Branch SNP, Blanton Forest SNP, and Kingdom Come SNP (Figure 9). Of the 95 small mammals collected in this study, 75 records (79%) were collected at Bad Branch SNP and Blanton Forest SNP (Table 5). Hi Lewis SNP had similar numbers of captures from each genus captured at the site (though Blarina was not captured), and contributed the most captures for Peromyscus and Microtus (Figure 9). Only specimens of Sorex were found at Kingdom Come SNP (Figure 9). Most small mammals were captured in May and June (Figure 10).

Small Mammal Community Composition

Canopy openness, the number of mountain laurel stems, the number of rocks, the leaf litter depth, and elevation were all statistically significant in explaining variation in relative abundances of small mammals at trap locations. The model including all of these variables explained 13.1% of the variation in small mammal community composition,
with axis 1 being best explained by the elevational gradient and axis 2 by the average number of mountain laurel stems rooted in quadrats (Figure 11). At the 95% significance level, average percent canopy openness (df = 1, F = 4.92, p = 0.01), average number of mountain laurel stems (df = 1, F = 3.47, p = 0.01), average number of rocks (df = 1, F = 4.01, p = 0.01), average leaf litter depth (df = 1, F = 3.15, p = 0.04), and elevation (df = 1, F = 4.60, p = 0.01) were found to vary significantly with relative abundance of small mammals.

*Sorex fumeus* and *Microtus pinetorum* had the highest correlation coefficients in the redundancy analysis (r = 0.99 and r = 0.97, respectively), and explained most of the variation in the data. Relative abundance of *S. fumeus* had the greatest amount of variation along the x-axis, and *M. pinetorum* had the greatest amount of variation along the y-axis (Figure 11). Relative abundance of *S. fumeus* was most strongly associated with elevation, with fewer *S. fumeus* found at higher elevations compared to lower elevations. *Sorex fumeus* abundance was also negatively associated with canopy openness and the number of mountain laurel stems rooted at trap locations. Additionally, relative abundance of *S. fumeus* was higher in areas with deeper leaf litter and higher numbers of rocks. *Microtus pinetorum* abundance was higher in areas with higher numbers of mountain laurel stems, greater canopy openness, and at higher elevations (Figure 11). There was also a negative association between relative abundance of *M. pinetorum* and average number of rocks at trap location, and a much weaker negative relationship with average leaf litter depth at trap location.

Relative abundances of all other species captured had correlation coefficients less than 0.2. Even so, relative abundance of *Peromyscus leucopus* was lower in areas with
deeper leaf litter and higher numbers of mountain laurel stems rooted at trap location (Figure 11). There was higher *P. leucopus* abundance in areas at higher elevations and with higher numbers of rocks. *Sorex cinereus* and *B. brevicauda* abundance was higher in areas with deeper leaf litter and higher numbers of rocks (Figure 11). Additionally, abundance of both species was lower in areas at higher elevations and with higher average canopy openness. *Peromyscus maniculatus, M. gapperi*, and *S. hoyi* had correlation coefficients so low they did not have discernable relationships with any environmental variables (Figure 11).

*Generalized Linear Mixed Models*

The most plausible model for small mammal species richness included the effects of soil temperature, number of rocks, and leaf litter depth at the trap location (Table 6). The model predicted a 76.8% decrease in species richness for each 1°C increase in soil temperature (Table 7). Additionally, the model predicted a 110.2% increase in species richness for every additional rock within the quadrat (Table 7). Of the eight additional competing models (Table 6), all included the effect of number of rocks at the trap location as a predictor variable. One model included average percent rhododendron cover, elevation, average duff depth, and average number of rhododendron stems rooted at the trap location as predictor variables. Three models included average soil temperature, average width of CWD, and average leaf litter depth at the trap location as predictor variables.

The most plausible model for total abundance of small mammals included the effects of percent rhododendron cover, number of rocks, and soil temperature at the trap location (Table 8). The model predicted a 69.2% decrease in total small mammal
abundance with each 1°C increase in soil and a 111.3% increase in total small mammal abundance with each additional rock at a trap location (Table 9). The model also predicted a 100.8% increase in total small mammal abundance with each additional percent increase in rhododendron cover (Table 9). Of the two competing models, both included the effects of number of rocks and soil temperature at a trap location (Table 8). One of the competing models included average leaf litter depth at trap location and the other included average width of CWD at trap location as predictor variables.

**Discussion**

**Trapping Effort**

Catch per unit effort can vary widely across studies of small mammal communities. Ford et al. (1999) reported a catch per unit effort of 1.91 per 100 TN after pitfall trapping 76,103 TN. McCay et al. (1998) reported a particularly high catch per unit effort of 7.20 per 100 TN over 6,720 TN. The trap success reported by Ford et al. (1999) and McCay et al. (1998) were much higher than what I observed (0.4 per 100 TN). Their higher trap success was likely due not only to the large number of trap nights, but also the number of pitfall traps deployed, i.e., Ford et al. (1999) deployed 20,946 pitfall traps compared to my deployment of 140 pitfall traps. McCay et al. (1998) suggested that trap success was likely a function of trapping scheme and targeted habitat characteristics. While the trapping scheme in this investigation was designed to maximize trap success, it is possible that more suitable habitats could have been targeted to further increase the number of small mammals captured. I record small mammal trapping activities conducted on Pine Mountain, in the future one could consider selecting trap sites on the basis of
forest floor structure and moisture; habitat characteristics which are reported to influence small mammal presence (McCay et al. 1998).

Microhabitat Conditions Affecting Small Mammal Community Composition

Elevation has been reported as influencing small mammal occurrence, and was likewise one of the strongest predictor variables in my investigation. I found *S. fumeus* in greatest abundance at lower elevation plots. Abundance of *S. fumeus* was reported to increase along a gradient of 245 m to 460 m ASL in Georgia’s Chattahoochee National Forest (Ford et al. 1994), and was trapped more commonly at high elevations (701 m and 1219 m ASL) on Big Black Mountain in Harlan County, Kentucky (Barbour 1951). Ford et al. (2005) also found an increase in abundance of *S. fumeus* between 246 m and 1600 m ASL in the southern Appalachian Mountains (Ford et al. 2005). I may not have observed an increase in *S. fumeus* abundance with increasing elevation because of the relatively narrow elevational range in my investigation. I did, however, find that *P. leucopus* was more abundant at higher elevations, which is consistent with previous investigations. I captured *P. leucopus* at plots ≤800 m ASL, which is in the range (548 m to 1264 m ASL) reported for populations on Big Black Mountain (Barbour 1951). Ford et al. (1994, 2005) reported a positive relationship between *S. cinereus* occurrence and elevation 245 m to 1600 m ASL. On Big Black Mountain, *S. cinereus* has exclusively been recorded between 1066 m to 1249 m ASL (Barbour 1951). This contrasts with my observation that the relative abundance of *S. cinereus* was higher at lower elevation plots.

On Pine Mountain, *M. pinetorum* was more abundant at higher elevation plots. Previous investigations trapped *M. pinetorum* between sea level and 600 m ASL in Virginia (Bellows et al. 2001), which is consistent with the elevation at which I captured
the species (>550 m ASL to ≤700 m ASL). While *Microtus pinetorum* has also been
recorded at elevations above 1300 m ASL in the southern Appalachian Mountains (Trani
et al. 2007), I did not capture any at my higher elevation plots (>700m). *Blarina*
brevicauda has been reported to inhabit an elevational range of 246 m to 1600 m ASL in
the southern Appalachian Mountains (Ford et al. 2005), with abundance increasing with
elevation (Ford et al. 1997, 2005). This contrasts with my observation that abundance of
*B. brevicauda* on Pine Mountain was higher at lower elevation plots. However, I captured
relatively few specimens from these groups and, while relative abundance for *M.*
pinetorum varied the most along second axis that corresponded to mountain laurel stems,
relative abundance of *B. brevicauda* varied little along either axis in my redundancy
analysis.

The number of mountain laurel stems was found to be an important predictor of
the relative abundance of small mammals on Pine Mountain. *Sorex cinereus* has been
recorded in areas with mountain laurel prevalent in the understory (Ford et al. 2001).
Although I observed slightly lower relative abundance of *S. cinereus* at sites with more
mountain laurel stems, *S. cinereus* often inhabits areas with a dense understory to prevent
dehydration (Whitaker 2004). I also found a slightly higher relative abundance of *S.*
fumeus in plots with more mountain laurel stems. *Sorex fumeus* is more commonly
described as being associated with CWD and rocky talus (Owen 1984, Trani et al. 2007),
rather than with understory vegetation. I may not have detected a relationship between
the relative abundance of *S. cinereus* and *S. fumeus* and mountain laurel stem density
because I had only seven *S. cinereus* specimens. Additionally, mountain laurel was only
present at one site, in low quantities compared with other understory species, where I
captured *S. cinereus*. Therefore, I think it likely that mountain laurel is not an important habitat feature for *Sorex* on Pine Mountain. In contrast, previous studies suggest a dense understory is an important habitat characteristic for *P. leucopus* (Kaufman et al. 1983). Although I observed that the relative abundance of *P. leucopus* did not vary with the number of mountain laurel stems, other understory species, e.g. rhododendron, may have provided the structure needed for *P. leucopus* on Pine Mountain.

I also found that relative abundance of *M. pinetorum* was higher in quadrats with more mountain laurel stems rooted at trap locations. *Microtus pinetorum* does use habitat with dense understory, but may prefer understories composed of the herbaceous plants that comprise their diet (Trani et al. 2007). In contrast, relative abundance of *B. brevicauda* did not vary with the number of mountain laurel stems rooted in quadrats in my investigation. This may be because, although *B. brevicauda* prefers habitat with a dense understory for the cover it provides, the vegetative composition of that understory does not appear to matter (George et al. 1986). Therefore, overall understory cover, rather than cover from a specific plant species, may be a better predictor for relative abundance of *B. brevicauda*.

Rocky debris is an important habitat feature for small mammals. *Sorex fumeus* abundance has been previously reported to be positively correlated with the amount of rocky debris (Owen 1984, McCay et al. 1998). Similarly, I found that abundance of *S. fumeus* was higher in quadrats with more rocks. I found the same relationship for *S. cinereus* and rocky debris. *Sorex cinereus* has also been reported to inhabit areas with more rocky debris (McCay et al. 1998, Whitaker 2004). *Sorex fumeus* and *S. cinereus* also use habitat with large amounts of CWD (Trani et al. 2007), making an overall
complex forest floor structure important for these species. However, habitat selection by *S. cinereus* is primarily driven by the need to avoid dehydration, rather than by the presence of rocks or CWD (Trani et al. 2007). The negative relationship I found between relative abundance of *P. leucopus* and average number of rocks at the trap location was weak, and therefore does not directly conflict with Kaufman et al. (1983), who did not observe a relationship between *P. leucopus* and rocky debris. However, *P. leucopus* occupies a wide variety of habitats, from forest to grassland, and has been reported to use habitat with rock features (Lackey et al. 1985).

In this study, the abundance of *M. pinetorum* was higher in areas with fewer rocks. *Microtus pinetorum*, overall, is not strongly associated with the presence of forest floor debris (Miller and Getz 1977). Understory cover (Trani et al. 2007) and leaf litter (Smolen 1981) appear to be more important habitat features, possibly because *M. pinetorum* consumes mostly herbaceous vegetation and uses leaf litter in the construction of their nests (Smolen 1981). *Blarina brevicauda* is also reported to not have a strong association with rocky debris (McCay et al. 1998). While I did observe higher relative abundance of *B. brevicauda* in quadrats with more rocks, there was overall low variation in relative abundance of *B. brevicauda* along the x-axis, the axis for which both abundance of *B. brevicauda* and average number of rocks at trap location had the greatest amount of variation. Relatively, there is some evidence that *B. brevicauda* will inhabit stone walls, due to the increased relative humidity around these rock features (George et al. 1986).

I found canopy openness to be an important predictor of small mammal community composition in Pine Mountain. Similar to what was noted in this study, Ford
et al. (1997) observed a decrease in the abundance of *S. fumeus* with increasing canopy openness. This may be because *S. fumeus* prefers habitat that provides shade and maintains consistent temperatures (Owen 1984). Ford et al. (1997) also found a positive, albeit insignificant, relationship between *S. cinereus* abundance and canopy openness. This contradicts my observation that relative abundance of this species tends to be lower in quadrats with greater canopy openness. *Sorex cinereus* does rely on cover to maintain water balance, but primarily from understory vegetation as opposed to the canopy (Whitaker 2004). While I did not observe a relationship between abundance of *P. leucopus* and canopy openness, population size has been reported to increase in response to increased canopy openness (Greenberg et al. 2006). This may be because *P. leucopus* prefers habitat with a dense understory for constructing nests, and an open canopy allows for the development of a dense understory (Lackey et al. 1985).

Ford et al. (1997) also found a positive, albeit insignificant, relationship between *B. brevicauda* abundance and canopy openness. This contradicts my observation that the relative abundance of this species tended to be lower in quadrats with greater canopy openness. This may be because *B. brevicauda* is tolerant of a range of local temperatures and does not typically rely on cover to act as a buffer against environmental conditions (George et al. 1986). In addition, I found that the relative abundance of *M. pinetorum* was higher in quadrats with greater canopy openness. *Microtus pinetorum* occupies forests with dense canopies and open grasslands (Smolen 1981), possibly because dense understory (Trani et al. 2007) and leaf litter (Smolen 1981) are believed to be more important habitat features for *M. pinetorum*. Although this does not support my finding that the relative abundance of *M. pinetorum* was positively correlated with canopy
openness, it may explain why I trapped *M. pinetorum* exclusively in my plots at Hi Lewis SNP, which was a site unique for having an understory primarily composed of ferns. Herbaceous vegetation is integral to the diet of *M. pinetorum* (Smolen 1981), possibly explaining why I only trapped *M. pinetorum* at Hi Lewis SNP.

Ford et al. (1997) found that the abundance of *S. fumeus* decreased with increasing leaf litter depth, while Brannon (2000) did not find leaf litter to be a significant predictor of the abundance of *S. fumeus*. Owen (1984), however, reported that *S. fumeus* tends to be found in areas with abundant leaf litter and burrows in leaf litter to help regulate body temperature. Similar to Ford et al (1994), I noted that *S. cinereus* was more abundant in quadrats with abundant leaf litter. This was likely because moisture is a limiting factor for *S. cinereus* (Whitaker 2004), and leaf litter can retain humidity. In contrast, I observed lower relative abundance of *P. leucopus* in quadrats with deeper leaf litter. *Peromyscus leucopus* is a habitat generalist, and has been recorded primarily in areas with complex vertical structure (Lackey et al. 1985). Therefore, leaf litter may not be an important habitat component for this species.

I observed lower relative abundance of *M. pinetorum* in quadrats with abundant leaf litter. Trani et al. (2007), however, reported that *M. pinetorum* abundance is positively correlated with leaf litter. *Microtus pinetorum* uses leaf litter for nesting material (Trani et al. 2007) and constructs its burrows beneath the leaf litter (Smolen 1981). However, *M. pinetorum* will inhabit areas with little leaf litter but a denser understory (Smolen 1981), which suggests the presence of cover, regardless of the form (leaf litter or canopy), is important as a habitat feature. Ford et al. (1997) found a positive relationship between abundance of *B. brevicauda* and leaf litter depth; similar to what
was noted in this study. Leaf litter may be vital to *B. brevicauda* because the animal burrows in leaf litter to maintain a constant body temperature and prevent dehydration (George et al. 1986).

Relative abundances of *P. maniculatus*, *M. gapperi*, and *S. hoyi* were not strongly correlated with any of the microhabitat variables included in the best fit model of my redundancy analysis. This is most likely because few specimens of these species were trapped throughout the field season. Previously, abundance of *P. maniculatus* in the Oregon Cascades was found to prefer habitat with greater canopy openness (Weldy et al. 2019) and shrubs in the understory (Trani et al. 2007). *Sorex hoyi* has also been found to prefer more open canopies, specifically in the southern Appalachian mountains (Ford et al. 1997). Otherwise, *S. hoyi* is a habitat generalist, and is not reported to be associated with specific microhabitat characteristics (Trani et al. 2007). Regarding elevational range, *P. maniculatus* has been trapped between 822 m and 914 m ASL in Harlan County, Kentucky (Barbour 1951) and between 245 m to 460 m ASL in the Chattahoochee National Forest (Ford et al. 1994). *Sorex hoyi* inhabits a wide range of elevations (160 m to 1600 m ASL) within the southern Appalachian Mountains (Ford et al. 2005). *Myodes gapperi* inhabits elevations ranging from 145 m to 460 m ASL in the Chattahoochee National Forest (Ford et al. 1994), and 685 m ASL on Big Black Mountain (Barbour 1951).

**Small Mammal Species Richness and Microhabitat Conditions**

Small mammal species richness on Pine Mountain was affected by many of the measured microhabitat conditions. Previous investigations, in other parts of the Americas, found a positive relationship between small mammal species richness and the
amount of leaf litter (Santos-Filho et al. 2016). Species richness of small mammals in the southern Appalachian Mountains has been found to increase in more mesic forest conditions (McShea et al. 2003). Soricid species richness has been found to increase with increasing ground temperatures (Hartling and Silva 2004), and elevation in the southern Appalachian Mountains (Ford et al. 2005). Previous investigations did not detect a relationship between small mammal species richness and number of rocks in Appalachian forests (Mitchell et al. 1997). However, I found that small mammal species richness on Pine Mountain decreased with increasing soil temperature, and increased with the average amount of rocky debris and leaf litter depth.

Total abundance of small mammals on Pine Mountain also varied with measured microhabitat conditions. Small mammal abundance increased with increasing ground temperature on Prince Edward Island (Hartling and Silva 2004) and elevation in the southern Appalachian Mountains (Ford et al. 1994). Additionally, small mammal abundance, particularly that of M. gapperi, has been found to be positively correlated with the amount of rocky debris (Orrock et al. 2000). The abundance of small mammals in the southern Appalachian mountains had also been reported to increase in more mesic forest conditions (McShea et al. 2003). On Pine Mountain, small mammal abundance increased with the average amount of rocky debris and percent rhododendron cover, and decreased with increasing soil temperature.

Future Directions

Despite the number of trap nights accumulated throughout the field season, sample sizes were relatively low for the species captured in this study. This recognizably biases my redundancy analysis in favor of S. fumeus, which had the largest sample size,
and therefore greatest relative abundance. Even so, low trap success is common in small mammal studies, especially those that target soricids (Mengak and Guynn 1987, McCay et al. 1998, Brannon 2002). While it is possible that continuous sampling within the same field sites from May through November of 2021 may have depopulated local small mammal communities, this is not evidenced in the literature. Previous studies have varied widely in the length of the field season, from one month out of the year for three years resulting in 11,486 TN (Ford 2000) to eight months resulting in 69,150 TN (Ford et al. 1994). None of these studies reported concerns of depopulating the small mammal communities they targeted. Additionally, a three month investigation found that catch per unit effort remained constant across the sampling period, regardless of length (Kirkland et al. 1998).

The low amount of variation explained by the constrained ordination in this investigation suggests that other environmental variables are contributing to variation in relative abundances of small mammals on Pine Mountain. Recognizing this, future efforts in this study system should investigate differences in relative abundances between Hi Lewis SNP, which is managed with fire, and other field sites that are not managed with fire. Hi Lewis SNP was last burned in 2000, but has a history of sporadic wildfires dating back to 1888 (Napier 2020). While I was unable to investigate the impact of fire management in this study due to an absence of replication, future use of fire on Pine Mountain by agencies could be coordinated in such a way as to allow for assessing the impacts of this management tool on small mammals. In addition, although elevation was found to be a predictor of relative abundance in this study, the elevational gradient in this investigation was only a difference of 321 m ASL between the highest and lowest
elevation plots. Future investigations at other sites (e.g., Black Mountain, Kentucky’s tallest mountain) could be conducted to determine if the positive relationship between elevation and relative abundances of small mammals noted in this study is consistent across a larger elevational gradient. Further small mammal investigations at other study areas are certainly merited given the need for contemporary inventory efforts across much of eastern Kentucky.
Chapter 2: Assessing and updating the Branley A. Branson Museum of Zoology's mammal collection at Eastern Kentucky University

Introduction

Natural history collections are an undervalued resource for cataloging extant and extinct biota. As of 1993, there were approximately 2.5 billion specimens within natural history collections, with a growth rate between 3-5% yearly (Duckworth et al. 1993). With as many as 30 million species globally (Duckworth et al. 1993), it is important to have a system for describing and cataloging the various species on Earth. Natural history collections and museums provide such a system, but often lack financial support and resources (Duckworth et al. 1993) because the public fails to see the value of research collections to which they do not have access (Allmon 1994). However, scientists from a variety of fields, ranging from molecular biology to biogeography, rely on natural history collections for research and educational purposes (Patterson 2002). Additionally, specimens are often also displayed for the enjoyment and education of the public (Patterson 2002).

There are many different types of specimens within natural history collections that provide data valuable to scientific investigations. Records can range from physical specimens to recordings and images (Duckworth et al. 1993), and are important for providing occurrence records. Museum specimens, and the information collected with them, are important for keeping record of overall presence and global distribution of biodiversity (Allmon 1994). Without documenting biodiversity, it is impossible to know what may be lost over time or in areas undergoing land use change (Allmon 1994). Additionally, natural history collections contain type and voucher specimens that are
critical for formally describing new species or confirming the identity of a specimen (Allmon 1994). These historic records can be used for documenting changes in community composition (Lister et al. 2011), and long-term studies of evolution (Holmes et al. 2016) and decline of populations within a given area (Shaffer et al. 1998). However, records must be updated with specimens from the same, or similar, areas in order for declines to be documented (Shaffer et al. 1998). Multiple updates over multiple years is ideal, as yearly fluctuations in populations can make detecting local declines difficult (Shaffer et al. 1998).

Despite public concerns, museum specimen collection generally does not impact small mammal populations. Controlled removal experiments found change in small mammal abundance and richness negligible among experimental removal and control communities (Hope et al. 2018). Compared to such systematic sampling, collection of specimens to add to museum collections is minimal and does not impact small mammal populations (Hope et al. 2018). Even for relatively small populations, or populations of unknown size, capture of small mammals can be justified as increasing what little knowledge may currently be available for that species (Ferguson 2020). Additionally, such sampling is being increasingly supplemented with digital records, such as images and recordings (Schindel and Cook 2018). Collecting records in this format not only increases accessibility (Schindel and Cook 2018), but may also reduce removal of small mammals from populations for collection purposes.

Natural history collections are important for describing new species and monitoring populations. For many natural history collections in the United States, the majority of mammal specimens were collected locally or from within the same state as
the collection (Howell 1923). Despite the many mammal specimens within collections (Howell 1923), new species are still being discovered (Patterson 2002). This may be in part because mammals can be difficult to observe, such as small mammals which are often reclusive and require trapping in order for them to be properly identified (Patterson 2002). Collection of these mammals then becomes important for documenting their presence. Mammal collections are also important for documenting local mammal diversity because repeated sampling of biodiversity within a local area can be used to document changes within the population over time (Shaffer et al. 1998).

The historical context provided by occurrence records is important for documenting species distributions. Museum collections document capture locations and standard measurements of specimens. However, these records must be updated to provide accurate data for long-term investigations. In North America, small mammal species richness is highest in the Appalachian and Cascade Mountains (Berman et al. 2007). An investigation of the literature revealed that rock shrews (Sorex dispar), Allegheny woodrats (Neotoma magister), southern bog lemmings (Synaptomys cooperi), woodland jumping mice (Napaeozapus insignis), meadow jumping mice (Zapus hudsonius), white-footed mice, and various shrew species are present in the Appalachian Mountains (Greenberg 2002, Ford et al. 2005, Campbell et al. 2010).

Beyond the peer-reviewed literature, another useful source of mammal occurrence records are species lists compiled by state and federal agencies. Management plans for nature preserves managed by the Office of Kentucky Nature Preserves (OKNP) report the presence of pygmy shrews (Sorex hoyi), woodland jumping mice, white-footed mice, deer mice (Peromyscus maniculatus), and smoky shrews (Sorex fumeus) on Pine
Mountain (Quinlan et al. 2007, 2009, Napier 2020) (Table 1). The most recent, small mammal survey on Pine Mountain was conducted in 1997 (Quinlan et al. 2007, 2009, Napier 2020). Renewed trapping efforts are necessary to assess the current small mammal diversity present on Pine Mountain for use in OKNP management plans.

The Branley A. Branson Museum of Zoology’s mammal collection at Eastern Kentucky University (hereafter, EKU mammal collection) contains biodiversity records for Kentucky, but has not recently acquired new small mammal specimens. The mammal collection at EKU moved to a new building in 2017, and the museum database has been updated to reflect specimens within the collection. This investigation aimed to survey small mammal communities at four nature preserves on Pine Mountain. My objective was to assess the status of records for reclusive small mammals within the EKU natural history collections and contribute to the collection through a systematic survey of small mammals on Pine Mountain in southeastern Kentucky.

Methods

Site Description

The Cumberland Mountains are located along Kentucky’s border with Virginia. Pine Mountain lies within the Level III Central Appalachian and Level IV Cumberland Mountain Thrust Block ecoregions (Woods et al. 2002). Pine Mountain stretches over 200 km from Whitley County to Pike County (McGrain 1983, Kleber and Kentucky Bicentennial Commission 1992) along the western side of the Cumberland Mountains (Braun 1935). Coal deposits and soils, developed from oceanic deposits during the Pennsylvanian and Mississippian Periods, characterize the geology of the Cumberland Mountains (McGrain 1983). On Pine Mountain, the soils have high sand content (Braun
Soil composition and vegetative communities vary at the microscale along the length of Pine Mountain (Braun 1935), contributing to the diversity of forests documented on Pine Mountain.

The Cumberland Mountains contain multiple, distinct forest types as a result of topography and climate. Rock formations in the region are primarily sandstone (Woods et al. 2002), such as the Lee and Pennington formations (Kleber and Kentucky Bicentennial Commission 1992). Average winter temperatures range from -6.7ºC to 6.7ºC, and average summer temperatures range from 16.7ºC to 30ºC (Woods et al. 2002). Forests in the Cumberland Mountains are primarily wet and humid, and average annual precipitation can exceed 140 cm (Woods et al. 2002). Mixed-mesophytic and mesic forests are common and contain a diversity of deciduous species (Woods et al. 2002). However, although mesic forests types are common, xeric forests are still important forest types (Braun 1935). The topography and climate surrounding Pine Mountain creates an interesting mosaic of forests.

I conducted my investigation at four state nature preserves (hereafter, SNPs) located in southeastern Kentucky on Pine Mountain: Blanton Forest, Bad Branch, Hi Lewis, and Kingdom Come SNPs. Bad Branch SNP and Kingdom Come SNP are located in Letcher County, Blanton Forest SNP in Harlan County, and Hi Lewis SNP in Harlan and Letcher Counties (Figure 1). All field sites used in this investigation are under the stewardship of the Office of Kentucky Nature Preserves (OKNP). The OKNP protects these natural areas because of their cultural and scientific value (Kleber and Kentucky Bicentennial Commission 1992). The approximately 1250 ha Blanton Forest SNP, located on the south-facing side of Pine Mountain, contains the largest amount of diverse
and old-growth forest found in Kentucky (Quinlan et al. 2009). Hi Lewis SNP study site is approximately 290 ha and contains pine-oak and pine barren forests (Napier 2020). Kingdom Come SNP is east of all field sites except Bad Branch SNP and provided access to the north-facing slope of Pine Mountain (Kentucky Energy and Environment Cabinet n.d.). Bad Branch SNP field site is approximately 1070 ha and is predominantly on the south-facing slope of Pine Mountain (Quinlan et al. 2007).

*Trapping Design*

I used pitfall traps to collect reclusive shrews and other small mammals within 20 m × 20 m plots stratified across all study sites. Plots were separated by at least 15 m within sites to ensure they were statistically independent (Findley and Yates 1991). Replicate plots were placed at low (≤550 m), medium (>550 m and ≤700 m), and high (>700 m) elevation areas, based on accessibility, at each field site (Table 3). I established mid and low elevation plots at Blanton Forest SNP ranging from 469 m to 616 m ASL, mid and high elevation plots at Hi Lewis SNP ranging from 617 m to 757 m ASL, high elevation plots at Kingdom Come SNP ranging from 789 m to 790 m ASL, and mid and low elevation plots at Bad Branch SNP ranging from 538 m to 641 m ASL (Table 3). I determined the elevation of each plot using a Garmin 64s GPS unit.

Trap success is typically low when trying to capture small mammal species (Mengak and Gunyn 1987, McCay et al. 1998, Brannon 2002). Trap type and location are therefore important when considering sample size and statistical analysis of data. Pitfall traps catch greater numbers and diversity of small mammals than do conventional small mammal traps (Williams and Braun 1983). Adding drift fences to pitfall trap arrays may increase trap success for certain small mammal groups, such as shrews (Williams and
Braun 1983). Using natural features (e.g., logs, rocks, and tree stumps) as drift fences has been reported to increase trap success (McCay et al. 1998). Within plots, I placed 10 pitfall traps [trap = 2 L plastic container (17 cm depth × 15 cm diameter)] next to natural drift fences (
Pitfall trapping was conducted May through November 2021; traps were checked bi-weekly in May and June, and then monthly thereafter. Holes were drilled around the rim of pitfall traps to enhance drainage and reduce flooding from rainfall. In addition, plastic plates were used to cover pitfall traps and prevent flooding (Figure 3). Debris was used to prop up the covers and allow small mammals to enter the traps. Pitfall traps were filled with 500 ml of 95% ethanol mixed with 500 ml of water, and enough mineral oil to cover the surface. Mineral oil was added to reduce evaporative loss and accelerate drowning of captured animals (Howard and Brock 1961, Sikes and Gannon 2011). This approach is consistent with ongoing pitfall trapping efforts currently used by the Kentucky Department of Fish and Wildlife Resources (KDFWR) as part of their small mammal survey efforts in the Cumberland Mountains (pers. comm., Zack Couch, Nongame Program Coordinator, KDFWR). Furthermore, these methods met with documented approval from the KDFWR, the OKNP, and Eastern Kentucky University’s (EKU) Institutional Animal Care and Use Committee (Protocol 02-2021).
**Specimen Preparation and Museum Curation**

All small mammal specimens collected were prepared according to the procedure outlined in Martin et al. (2011) and deposited in the EKU mammal collection (KDFWR Educational Collection Permit # SC2011236, EKU IACUC Protocol 02-2021). Degraded samples from pitfall traps were skeletonized. For intact specimens, I collected all standard measurements (total length, tail length, hind-foot length, ear length, weight) prior to skeletonizing, to prevent introducing inaccuracies from skin deformation during skinning (Martin et al. 2011). I kept all specimens within preserving fluid (70% EtOH) to preserve specimens and wore nitrile gloves during specimen processing to reduce likelihood of disease transmission. I weighed and sexed specimens, where possible, by examining for testes and nipples during dissection (Martin et al. 2011). Specimens were identified using resources provided by Dr. Charles Elliott and standard measurements from historic records of small mammals from eastern Kentucky [James Kiser (Stantec Consulting Service, Inc.) and John MacGragor (KDFWR)] were consulted to help with species identifications (Figure 12).

**Statistical Analyses**

The coding program R was used for all statistical analysis and data visualization (R Core Team 2019). The average rate of record acquisition for specimens collected in Kentucky was visualized using the ggplot2 package (Wickham 2016 p. 2), and rworldmap (South 2011) and dplyr version 1.0.3 (Wickham et al. 2021) were used to visualize distribution of small mammal records in Kentucky. The average (± std. dev.) standard measurements were calculated for each species captured. These calculations exclude two specimens that could only be identified to genus. One specimen, identified
as *Sorex*, was missing the skull so standard measurements, excluding total length, were used in identification. The other specimen, identified as *Microtus*, was a skull and there were no standard measurements or other identifiable features to help identify the specimen to species.

**Results**

*Summary of Small Mammal Records in the Branley A. Branson Museum of Zoology*

The EKU mammal collection was established in 1955 with five white-footed mice specimens. Prior to this investigation, the collection contained 1,123 records of 92 different species from four countries. There were 493 small mammal records collected within Kentucky in the EKU mammal collection prior to this investigation (Figure 13). Additionally, while many records in the EKU mammal collection have similar collectors, the context of their collection are not listed in the database. Many records were added by students in mammalogy courses taught at EKU, but there is insufficient information within the museum database to determine if specimen acquisition was primarily part of systematic surveys or opportunistic collection.

Currently, there are 665 small mammal records within EKU’s collection. Of these records, 442 are from Kentucky, with only three records from Letcher County and 10 from Harlan County (Figure 14). However, records from the gray literature suggest that at least seven genera are present in these counties (Figure 15). The most recent addition to the collection, prior to this investigation, was a woodland vole (*Microtus pinetorum*) in 2017; the capture location was not specified. The most recent addition of 10 or more specimens, prior to this investigation, was in 1999. All of these specimens were collected
in Madison County, Kentucky. Only three specimens in the collections are recorded as collected from Pine Mountain.

There are 14 specimens in the EKU small mammal collection that were recorded as having been collected in Harlan and Letcher counties, with 10 collected in Harlan County and four collected in Letcher County. Of the records from Harlan County, three are P. leucopus, four are Myodes gapperi, and one record each for Blarina brevicauda, Sorex cinereus, and P. maniculatus. Of the records from Letcher County, there is one record for each P. leucopus, Tamias striatus, M. gapperi, and Neotoma floridana. The earliest records date to 1972 and the most recent to 1985.

Summary of Trapping Effort

In total, survey efforts yielded 24,472 trap nights (TN; a single night over which a trap was deployed and operable) across all sites, including 7,550 TN at Bad Branch SNP, 7,750 TN at Blanton Forest SNP, 7,381 TN at Hi Lewis SNP, and 1,791 TN at Kingdom Come SNP. A total of 94 traps were disturbed throughout the field season. These traps were removed from their original location, likely by raccoons, and were considered closed for the previous trapping period. Of these, 10 traps were disturbed at Bad Branch SNP, 15 at Blanton Forest SNP, 19 at Hi Lewis SNP, and 50 Kingdom Come SNP. Most trap disturbances occurred in May through June and September through October (Figure 8).

Ninety-five small mammals were captured, yielding an overall catch per unit effort of 0.4 per 100 TN. Catch per unit effort across sites were as follows: 0.6 for Bad Branch SNP, 0.4 for Blanton Forest SNP, 0.2 for Hi Lewis SNP, and 0.3 for Kingdom Come SNP. Captures across genera included 74 Sorex, 10 Peromyscus, five Microtus,
four *Blarina*, and two *Myodes*. Eight small mammal species within these genera were captured, 68% of which were *S. fumeus* (Error! Reference source not found.). One specimen each was collected for *S. hoyi* and *P. maniculatus* (Error! Reference source not found.). One specimen could only be identified to genus, because it was missing a skull. Standard measurements, excepting total body length, were used to identify the specimen. Within these genera, specimens from eight different species were captured (Error! Reference source not found., Error! Reference source not found.). Most small mammals were captured within May and June (Figure 10).

**Discussion**

*Small Mammal Records in the Branley A. Branson Museum of Zoology*

This investigation is the largest individual contribution of small mammal records to the Branley A. Branson Zoology Collection to-date. The records are also a part of a systematic study, whereas the origin of many small mammal records in the collection are from opportunistic collection. Updating natural history records is important for creating a database that can be used to assess long-term changes in species abundance (Shaffer et al. 1998). Long-term investigations can be used to determine if populations are declining (Shaffer et al. 1998).

This investigation added 95 specimens to the EKU small mammal collection, including the first specimens of *S. fumeus* and *S. hoyi* for both Harlan and Letcher counties. *Sorex fumeus* and *S. hoyi* had previously been reported to be present on Pine Mountain at Blanton Forest SNP (Quinlan et al. 2009), Bad Branch SNP (Quinlan et al. 2007), and Hi Lewis SNP (Napier 2020) from a systematic survey dating to 1997. Therefore, in order to track species presence at these field sites long-term, continued
survey efforts will be necessary to accurately catalog change in these communities over time.

During this study, eight species known to be present on Pine Mountain were captured. *Blarina brevicauda, M. gapperi, P. leucopus, S. cinereus, and S. fumeus* were captured at all sites where they were identified as present during the 1997 survey (Quinlan et al. 2007, 2009, Napier 2020). *Microtus pinetorum* was captured at Hi Lewis SNP, but previously was documented only at Bad Branch SNP (Quinlan et al. 2007). *Peromyscus maniculatus* was captured at Bad Branch SNP; it had previously been documented at Blanton Forest SNP (Quinlan et al. 2009) and Hi Lewis SNP (Napier 2020) in addition to Bad Branch SNP (Quinlan et al. 2007). *Sorex hoyi* was only captured at Hi Lewis SNP, but was previously documented at Blanton Forest SNP (Quinlan et al. 2009) and Bad Branch SNP (Quinlan et al. 2007) in addition to Hi Lewis SNP (Napier 2020). There are also small mammal species reported to have been trapped on Pine Mountain that were not captured during the field season. *Napaeozapus insignis* (Quinlan et al. 2007), *Parascalops breweri* (Quinlan et al. 2007, 2009), *Sorex dispar* (Quinlan et al. 2007), and *Synaptomys cooperi* (Quinlan et al. 2007, 2009) are known to be present at one or more field sites, but were not captured during this investigation.

**Limitations of Study and Future Directions**

Some limitations of this study include exclusively pitfall trapping within plots of similar habitat type. Some species, such as *S. dispar* (Trani et al. 2007), are habitat specialists and will only be captured along specific habitat features. Notably, I didn’t capture *S. dispar* but this species has been captured historically on Pine Mountain (Harker et al. 1979, Campbell et al. 1993), and at Bad Branch SNP specifically (Quinlan
et al. 2007). Future investigations should target species, like *S. dispar*, to determine the current population status on Pine Mountain. Additionally, while pitfall trapping has relatively high trap success, some species are more likely to be caught in snap traps (Mengak and Guynn 1987). Therefore, varying the trap types used may have yielded a higher diversity of fauna.

There are several small mammal species known to be present at the targeted field sites that were not captured in the pitfall traps. There are records of *Napaeozapus insignis* at Bad Branch SNP and Hi Lewis SNP (Quinlan et al. 2007, Napier 2020). *Neotoma floridana* has been captured at Blanton Forest SNP, Bad Branch SNP, and Hi Lewis SNP (Quinlan et al. 2007, 2009, Napier 2020). *Parascalops breweri* and *S. cooperi* have been recorded at both Blanton Forest SNP and Bad Branch SNP (Quinlan et al. 2007, 2009). There are also records of *S. dispar* present at Bad Branch SNP (Quinlan et al. 2007). These records date to 1997, so targeted trapping efforts for these species would be valuable for further updating of inventory records for small mammals on Pine Mountain.
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Appendix A: Figures
Appendix A: Figures

Figure 1. Map of Pine Mountain and field sites from west to east: Blanton Forest State Nature Preserve (SNP), Hi Lewis SNP, Kingdom Come SNP, and Bad Branch SNP. Harlan County borders Pine Mountain to the east, and Letcher County borders Pine Mountain to the west. The shapefile of SNP boundaries used in this map was provided by the Office of Kentucky Nature Preserves. The terrain background map is from map tiles by Stamen Design, under CC BY 3.0 Data by OpenStreetMap, under ODbL.
Figure 2. Pitfall trapping scheme. Pitfall traps (black circles) were placed alongside natural drift fences, such as logs (green cylinders), tree stumps (blue circle), and rocks (gray triangles). Canopy closure was measured in the center of 1 m$^2$ quadrats (blue, dashed squares), and leaf litter depth, duff depth, soil temperature, and soil moisture measurements were measured at every corner of each quadrat (as indicated by orange “X”s). This is not a full plot, and is only meant to demonstrate how traps were placed relative to habitat features.
Figure 3. Reference images of (A) an open trap covered by a plate and branches for weight with a rock as the natural drift fence, (B) an uncovered trap, and (C) a plot at Blanton Forest State Nature Preserve.
Figure 4. Boxplots showing the median and interquartile ranges of variation in rocky and coarse woody debris within 1 m² quadrats surrounding pitfall traps set at Bad Branch State Nature Preserve (SNP), Blanton Forest SNP, Hi Lewis SNP, and Kingdom Come SNP in eastern Kentucky.
Figure 5. Boxplot showing the median and interquartile ranges of variation in canopy openness measured within 1 m² quadrats surrounding pitfall traps set at Bad Branch State Nature Preserve (SNP), Blanton Forest SNP, Hi Lewis SNP, and Kingdom Come SNP in eastern Kentucky.
Figure 6. Boxplots showing the median and interquartile ranges of variation in understory cover data collected within 1 m² quadrats surrounding pitfall traps set at Bad Branch State Nature Preserve (SNP), Blanton Forest SNP, Hi Lewis SNP, and Kingdom Come SNP in eastern Kentucky.
Figure 7. Boxplots showing the median and interquartile ranges of variation in (A) duff depth, (B) leaf litter depth, (C) soil saturation, and (D) soil temperature collected within 1 m² quadrats surrounding pitfall traps set at Bad Branch State Nature Preserve (SNP), Blanton Forest SNP, Hi Lewis SNP, and Kingdom Come SNP in eastern Kentucky.
Figure 8. Trap disturbances throughout 2021 field season across Blanton Forest State Nature Preserve (SNP), Bad Branch SNP, Hi Lewis SNP, and Kingdom Come SNP in eastern Kentucky.
Figure 9. Total number of small mammals captured, by genus, throughout 2021 field season across Bad Branch State Nature Preserve (SNP), Blanton Forest SNP, Hi Lewis SNP, and Kingdom Come SNP in eastern Kentucky.
Figure 10. Number of small mammals captured throughout 2021 field season across Blanton Forest State Nature Preserve (SNP), Bad Branch SNP, Hi Lewis SNP, and Kingdom Come SNP in eastern Kentucky.
Figure 11. Biplot displaying results of redundancy analysis with Hellinger transformed abundance data for small mammals (S_fumeus = Sorex fumeus, S_cinereus = Sorex cinereus, M_pintrm = Microtus pinetorum, B_brevcd = Blarina brevicauda, M_gapper = Myodes gapperi, S_hoyi = Sorex hoyi, P_leucps = Peromyscus leucopus, and P_mnclts = Peromyscus maniculatus) captured along the ridge of Pine Mountain as the response variable and microhabitat conditions as predictor variables. The model included only microhabitat conditions that explained the largest amount of variation in the data [avg_canopy openness = average canopy openness (%), avg_num_mtnl_rooted = average number of mountain laurel stems rooted, avg_num_rd = average number of rocks, avg_leaf_litter_dep_cm = average leaf litter depth (cm), and Elevation (m) = elevation (m ASL) at trap location].
Figure 12. Summary of standard measurements used to confirm specimen identification of *Sorex dispar*, *Sorex cinereus*, *Sorex fumeus*, and *Sorex hoyi*. Standard measurements from Trani et al. (2007), Barbour and Davis (1974), and data provided by John MacGregor (Biologist, Kentucky Department of Fish and Wildlife Resources, Frankfort, KY).
Figure 13. Rate of acquisition of small mammal records from Kentucky, within the Eastern Kentucky University mammal collection, from 1955 to 2021.
**Figure 14.** Number of small mammal records in the Eastern Kentucky University mammal collection by genus and county in Kentucky, USA.
Figure 15. Number of species by genus within Clay, Harlan, Knox, Leslie, Letcher, Owsley, and Perry Counties in Kentucky, USA. These records are from Quinlan et al. (2007, 2009), Napier (2020), and Campbell et al. (1993).
Figure 16. Reference skulls from voucher specimens collected from pitfall trapping in Bad Branch State Nature Preserve (SNP), Blanton Forest SNP, Hi Lewis SNP, and Kingdom Come (SNP), eastern Kentucky.
Appendix B: Tables
Appendix B: Tables

Table 1. Species list of small mammals present at the Daniel Boone National Forest (DBNF), Cumberland Gap National Historic Park (CG), Cumberland Mountains (CM), Big Black Mountain (BBM), and Hi Lewis (HL), Bad Branch (BB), and Blanton Forest (BF) State Nature Preserves (SNP). Also included are trapping efforts along Pine Mountain (PM) within Clay, Leslie, and Harlan counties. Occurrence records for the DBNF are from Campbell et al. (1993), records for PM are from Campbell et al. (1993) and Harker et al. (1979), and occurrence records for CG and BBM are from Harker et al. (1979). SNP records are from management plans prepared by Napier (2020), and Quinlan et al. (2007, 2009).

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<td>BB</td>
<td>BB</td>
<td>BB, BF</td>
<td>BB, BF</td>
</tr>
<tr>
<td>Napaeozapus insignis</td>
<td>PM</td>
<td>BB</td>
<td>BB</td>
<td>BB</td>
<td>BB, BF</td>
<td>BB, BF</td>
</tr>
<tr>
<td>Neotoma floridana</td>
<td>DBNF</td>
<td>BB</td>
<td>BB</td>
<td>BB</td>
<td>BB, BF</td>
<td>BB, BF</td>
</tr>
<tr>
<td>Parascalops breweri</td>
<td>BB</td>
<td>BB</td>
<td>BB</td>
<td>BB</td>
<td>BB, BF</td>
<td>BB, BF</td>
</tr>
<tr>
<td>Peromyscus leucopus</td>
<td>PM</td>
<td>BB</td>
<td>BB</td>
<td>BB</td>
<td>BB, BF</td>
<td>BB, BF</td>
</tr>
<tr>
<td>Peromyscus maniculatus</td>
<td>PM, BBM</td>
<td>PM</td>
<td>BB</td>
<td>BB</td>
<td>BB, BF</td>
<td>BB, BF</td>
</tr>
<tr>
<td>Sorex cinereus</td>
<td>PM, BBM, CG</td>
<td>DBNF, PM</td>
<td>BB</td>
<td>BB</td>
<td>BB, BF</td>
<td>BB, BF</td>
</tr>
<tr>
<td>Sorex dispar</td>
<td>PM</td>
<td>DBNF, PM</td>
<td>BB</td>
<td>BB</td>
<td>BB, BF</td>
<td>BB, BF</td>
</tr>
<tr>
<td>Sorex fumeus</td>
<td>PM</td>
<td>BB</td>
<td>BB</td>
<td>BB</td>
<td>BB, BF</td>
<td>BB, BF, HL, KC</td>
</tr>
<tr>
<td>Sorex hoyi</td>
<td>PM</td>
<td>DBNF, PM</td>
<td>BB</td>
<td>BB</td>
<td>BB, BF</td>
<td>BB, BF</td>
</tr>
<tr>
<td>Synaptomys cooperi</td>
<td>PM</td>
<td>BB</td>
<td>BB</td>
<td>BB</td>
<td>BB, BF</td>
<td>BB, BF</td>
</tr>
<tr>
<td>Ochrotomys nuttalli</td>
<td>PM</td>
<td>BB</td>
<td>BB</td>
<td>BB</td>
<td>BB, BF</td>
<td>BB, BF</td>
</tr>
<tr>
<td>Reithrodontomys humulis</td>
<td>PM</td>
<td>BB</td>
<td>BB</td>
<td>BB</td>
<td>BB, BF</td>
<td>BB, BF</td>
</tr>
</tbody>
</table>
Table 2. List of species likely to be present on Pine Mountain, southeastern Kentucky, based on species distributions and habitat preference described by Barbour and Davis (1974).

<table>
<thead>
<tr>
<th>Scientific name</th>
<th>Common name</th>
<th>Habitat Preference</th>
</tr>
</thead>
<tbody>
<tr>
<td><em>Blarina brevicauda</em></td>
<td>Northern short-tailed shrew</td>
<td>Early successional habitat; Mesic forests</td>
</tr>
<tr>
<td><em>Myodes gapperi</em></td>
<td>Red-backed vole</td>
<td>Mesic forest</td>
</tr>
<tr>
<td><em>Cryptotis parva</em></td>
<td>Least shrew</td>
<td>Early successional habitat</td>
</tr>
<tr>
<td><em>Mus musculus</em></td>
<td>House mouse</td>
<td>Early successional habitat</td>
</tr>
<tr>
<td><em>Napaeozapus insignis</em></td>
<td>Woodland jumping mouse</td>
<td>Mesic forests; mid-successional habitat</td>
</tr>
<tr>
<td><em>Neotoma floridana</em></td>
<td>Eastern woodrat</td>
<td>Clifflines; rocky areas</td>
</tr>
<tr>
<td><em>Ochrotomys nuttalli</em></td>
<td>Golden mouse</td>
<td>Early to mid- successional habitat</td>
</tr>
<tr>
<td><em>Parascalops breweri</em></td>
<td>Hairy-tailed mole</td>
<td>Habitat generalist</td>
</tr>
<tr>
<td><em>Peromyscus leucopus</em></td>
<td>White-footed mouse</td>
<td>Mesic forests</td>
</tr>
<tr>
<td><em>Peromyscus maniculatus nubiterrae</em></td>
<td>Cloudland deer mouse</td>
<td>Habitat generalist</td>
</tr>
<tr>
<td><em>Microtus pinetorum</em></td>
<td>Pine vole</td>
<td>Developed areas</td>
</tr>
<tr>
<td><em>Rattus norvegicus</em></td>
<td>Norway rat</td>
<td>Early successional habitat</td>
</tr>
<tr>
<td><em>Reithrodontomys humulis</em></td>
<td>Eastern harvest mouse</td>
<td>Mesic forests; presence of coarse woody debris</td>
</tr>
<tr>
<td><em>Sorex cinereus</em></td>
<td>Masked shrew</td>
<td>Mesic forests; presence of rocky or coarse woody debris</td>
</tr>
<tr>
<td><em>Sorex fumeus</em></td>
<td>Smoky shrew</td>
<td>Early successional habitat</td>
</tr>
<tr>
<td><em>Synaptomys cooperi</em></td>
<td>Southern bog lemming</td>
<td>Mesic forests; presence of rocky or coarse woody debris</td>
</tr>
</tbody>
</table>
Table 3. Summary of accessible elevations at Pine Mountain field sites from west to east: Blanton Forest State Nature Preserve (SNP), Hi Lewis SNP, Kingdom Come SNP, and Bad Branch SNP. Accessible elevations are marked with an “X” and direction of sites relative to one another is indicated by the arrows at the bottom of the figure.

<table>
<thead>
<tr>
<th>Accessible Elevation</th>
<th>Blanton Forest SNP</th>
<th>Hi Lewis SNP</th>
<th>Kingdom Come SNP</th>
<th>Bad Branch SNP</th>
</tr>
</thead>
<tbody>
<tr>
<td>High (&gt;700m)</td>
<td>X</td>
<td>X</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Mid (&gt;550m; ≤700m)</td>
<td>X</td>
<td>X</td>
<td></td>
<td>X</td>
</tr>
<tr>
<td>Low (≤550m)</td>
<td>X</td>
<td></td>
<td></td>
<td>X</td>
</tr>
</tbody>
</table>

West ← → East
Table 4. Summary of small mammal species recorded across elevations at Pine Mountain field sites. Species are coded: 1 = *Sorex fumeus*, 2 = *Sorex cinereus*, 3 = *Peromyscus leucopus*, 4 = *Peromyscus maniculatus*, 5 = *Microtus pinetorum*, 6 = *Blarina brevicauda*, 7 = *Sorex hoyi*, and 8 = *Myodes gapperi*. Two specimens are omitted from the table, because they could only be identified to genus.

<table>
<thead>
<tr>
<th>Accessible Elevation</th>
<th>Blanton Forest SNP</th>
<th>Hi Lewis SNP</th>
<th>Kingdom Come SNP</th>
<th>Bad Branch SNP</th>
</tr>
</thead>
<tbody>
<tr>
<td>High (&gt;700m)</td>
<td>3, 5</td>
<td>1</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Mid (&gt;550m; ≤700m)</td>
<td>1, 2, 8</td>
<td>1, 3, 5, 7</td>
<td>1, 2, 3, 6</td>
<td></td>
</tr>
<tr>
<td>Low (≤550m)</td>
<td>1, 2, 6</td>
<td></td>
<td>1, 3, 4, 6, 8</td>
<td></td>
</tr>
<tr>
<td>West ←</td>
<td>→ East</td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>
Table 5. Summary of small mammals captured across elevations at Pine Mountain field sites. The total number of small mammals captured at each elevation across sites is recorded as the sum, and the percent of all records from each elevation across sites is recorded as composition.

<table>
<thead>
<tr>
<th>Accessible Elevation</th>
<th>Blanton Forest SNP</th>
<th>Hi Lewis SNP</th>
<th>Kingdom Come SNP</th>
<th>Bad Branch SNP</th>
</tr>
</thead>
<tbody>
<tr>
<td>High (&gt;700m)</td>
<td>3 (3.2)</td>
<td>6 (6.3)</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Mid (&gt;550m; ≤700m)</td>
<td>13 (13.7)</td>
<td>11 (11.6)</td>
<td>32 (33.7)</td>
<td></td>
</tr>
<tr>
<td>Low (≤550m)</td>
<td>17 (17.9)</td>
<td></td>
<td></td>
<td>13 (13.7)</td>
</tr>
</tbody>
</table>

West ←  East
Table 6. Results of AICc model selection for generalized linear mixed models of species richness of small mammals captured on Pine Mountain. Predictor variables include elevation (m above sea level), average width of coarse woody debris (CWD; cm), average duff depth (cm), average leaf litter depth (cm), average number of rocks, average number of rhododendron stems rooted, percent rhododendron cover, and average soil temperature (°C) measured within quadrats.

<table>
<thead>
<tr>
<th>Predictor Variables</th>
<th>df</th>
<th>AIC&lt;sub&gt;c&lt;/sub&gt;</th>
<th>ΔAIC&lt;sub&gt;c&lt;/sub&gt;</th>
<th>w&lt;sub&gt;i&lt;/sub&gt;</th>
</tr>
</thead>
<tbody>
<tr>
<td>Avg. litter depth + avg. number of rocks + avg. soil temp.</td>
<td>5</td>
<td>235.79</td>
<td>0.00</td>
<td>0.21</td>
</tr>
<tr>
<td>Avg. number of rhodo + % rhodo cover + avg. soil temp.</td>
<td>5</td>
<td>236.72</td>
<td>0.93</td>
<td>0.13</td>
</tr>
<tr>
<td>Avg. CWD width + avg. number of rocks + avg. soil temp.</td>
<td>5</td>
<td>237.02</td>
<td>1.23</td>
<td>0.11</td>
</tr>
<tr>
<td>Elevation + avg. litter depth + avg. number of rocks</td>
<td>5</td>
<td>237.04</td>
<td>1.25</td>
<td>0.11</td>
</tr>
<tr>
<td>Avg. number of rocks + avg. soil temp.</td>
<td>4</td>
<td>237.37</td>
<td>1.59</td>
<td>0.10</td>
</tr>
<tr>
<td>Avg. CWD width + avg. number of rocks</td>
<td>4</td>
<td>237.48</td>
<td>1.70</td>
<td>0.09</td>
</tr>
<tr>
<td>Avg. number of rocks + avg. number rhodo + avg. soil temp.</td>
<td>5</td>
<td>237.64</td>
<td>1.86</td>
<td>0.08</td>
</tr>
<tr>
<td>Avg. duff depth + avg. litter depth + avg. number of rocks</td>
<td>5</td>
<td>237.67</td>
<td>1.88</td>
<td>0.08</td>
</tr>
<tr>
<td>Avg. CWD width + avg. litter depth + avg. number of rocks</td>
<td>5</td>
<td>237.72</td>
<td>1.93</td>
<td>0.08</td>
</tr>
</tbody>
</table>
Table 7. Parameters of predictor variables for species richness in the best fit generalized linear mixed model of small mammals captured on Pine Mountain.

<table>
<thead>
<tr>
<th>Parameter</th>
<th>Estimate</th>
<th>Standard Error</th>
<th>95% CI</th>
</tr>
</thead>
<tbody>
<tr>
<td>Intercept</td>
<td>2.25</td>
<td>2.09</td>
<td>-1.84, 6.34</td>
</tr>
<tr>
<td>Average Soil Temperature (°C)</td>
<td>-0.26</td>
<td>0.13</td>
<td>-0.52, -0.01</td>
</tr>
<tr>
<td>Average Number of Rocks</td>
<td>0.10</td>
<td>0.03</td>
<td>0.05, 0.15</td>
</tr>
<tr>
<td>Average Leaf Litter Depth (cm)</td>
<td>0.23</td>
<td>0.12</td>
<td>-0.001, 0.46</td>
</tr>
</tbody>
</table>
Table 8. Results of AICc model selection for generalized linear mixed models of total abundance of small mammals captured on Pine Mountain. Predictor variables include average width of coarse woody debris (CWD, cm), average leaf litter depth (cm), average number of rocks, percent rhododendron cover, and average soil temperature (°C) within quadrats.

<table>
<thead>
<tr>
<th>Predictor Variables</th>
<th>df</th>
<th>AICc</th>
<th>ΔAICc</th>
<th>wi</th>
</tr>
</thead>
<tbody>
<tr>
<td>Avg. number of rocks + % rhodo cover + avg. soil temp.</td>
<td>5</td>
<td>301.82</td>
<td>0.00</td>
<td>0.48</td>
</tr>
<tr>
<td>Avg. litter depth + avg. number of rocks + avg. soil temp.</td>
<td>5</td>
<td>302.59</td>
<td>0.76</td>
<td>0.33</td>
</tr>
<tr>
<td>Avg. CWD width + avg. number of rocks + avg. soil temp.</td>
<td>5</td>
<td>303.72</td>
<td>1.90</td>
<td>0.19</td>
</tr>
</tbody>
</table>
Table 9. Parameters of predictor variables for total abundance in the best fit generalized linear mixed model of small mammals captured on Pine Mountain.

<table>
<thead>
<tr>
<th>Parameter</th>
<th>Estimate</th>
<th>Standard Error</th>
<th>95% CI</th>
</tr>
</thead>
<tbody>
<tr>
<td>Intercept</td>
<td>4.88</td>
<td>1.90</td>
<td>1.15, 8.61</td>
</tr>
<tr>
<td>Average Number of Rocks</td>
<td>0.11</td>
<td>0.02</td>
<td>0.07, 0.15</td>
</tr>
<tr>
<td>Average % Rhododendron Cover</td>
<td>0.01</td>
<td>0.002</td>
<td>0.003, 0.01</td>
</tr>
<tr>
<td>Average Soil Temperature (ºC)</td>
<td>-0.37</td>
<td>0.12</td>
<td>-0.60, -0.13</td>
</tr>
</tbody>
</table>
Table 10. Average standard measurements (± standard deviation) for all specimens collected in this study. Error is not included for *Sorex hoyi* and *Peromyscus maniculatus* given they are singular records. Two specimens could not be identified to species because they lacked sufficient identifying characteristics; one of which is not included in this table because only the skull was present.

<table>
<thead>
<tr>
<th>Species</th>
<th># Collected (%) of total</th>
<th>Hind Foot Length (mm)</th>
<th>Tail Length (mm)</th>
<th>Total Length (mm)</th>
<th>Ear Length (mm)</th>
<th>Weight (g)</th>
</tr>
</thead>
<tbody>
<tr>
<td><em>Blarina brevicauda</em></td>
<td>4 (4.2)</td>
<td>14.2 ± 0.3</td>
<td>23 ± 0.8</td>
<td>107.5 ± 5.5</td>
<td>2.5 ± 0.7</td>
<td>5.6 ± 6.5</td>
</tr>
<tr>
<td><em>Sorex hoyi</em></td>
<td>1 (1.1)</td>
<td>8.5</td>
<td>24</td>
<td>69</td>
<td>3</td>
<td>2</td>
</tr>
<tr>
<td><em>Sorex cinereus</em></td>
<td>7 (7.4)</td>
<td>12.1 ± 1.0</td>
<td>40.9 ± 1.7</td>
<td>100.9 ± 4.1</td>
<td>4.0 ± 0.6</td>
<td>3.6 ± 2.7</td>
</tr>
<tr>
<td><em>Sorex fumeus</em></td>
<td>65 (68.4)</td>
<td>12.8 ± 0.6</td>
<td>41.0 ± 2.5</td>
<td>100.2 ± 5.2</td>
<td>4.3 ± 0.9</td>
<td>5.8 ± 2.2</td>
</tr>
<tr>
<td><em>Microtus pinetorum</em></td>
<td>4 (4.2)</td>
<td>14.8 ± 0.5</td>
<td>17.6 ± 1.3</td>
<td>99.1 ± 7.1</td>
<td>6.8 ± 0.5</td>
<td>8.2 ± 11.3</td>
</tr>
<tr>
<td><em>Myodes gapperi</em></td>
<td>2 (2.1)</td>
<td>17.8 ± 0.4</td>
<td>24.5 ± 0.7</td>
<td>96 ± 5.7</td>
<td>7.5 ± 0.7</td>
<td>7.9 ± 7.6</td>
</tr>
<tr>
<td><em>Peromyscus leucopus</em></td>
<td>9 (9.5)</td>
<td>17.1 ± 5.8</td>
<td>63.1 ± 14.2</td>
<td>126.9 ± 25.9</td>
<td>14.1 ± 1.9</td>
<td>12.8 ± 10.4</td>
</tr>
<tr>
<td><em>Peromyscus maniculatus</em></td>
<td>1 (1.1)</td>
<td>19</td>
<td>80</td>
<td>148</td>
<td>14</td>
<td>14.8</td>
</tr>
<tr>
<td>Unknown specimen #1</td>
<td>1 (1.1)</td>
<td>13</td>
<td>44</td>
<td>NA</td>
<td>NA</td>
<td>NA</td>
</tr>
</tbody>
</table>