FINAL REPORT

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Project Title: Assessing occurrence and habitat associations of six priority small mammal species in high elevation Appalachian forests of South Carolina

Accomplishments: The project was completed on time and resulted in a Masters Thesis by Katelyn N. Steen entitled, "Habitat Associations of Small Mammals and Their Role as Sentinels for Biodiversity in the Southern Blue Ridge." This thesis is appended here as the final report for this grant.

Summary:

There are approximately 40 small mammals in the Blue Ridge Ecoregion of South Carolina excluding bats, and 11 are priority species for conservation in the SC SWAP. Due to the high cost and labor investment of traditional small mammal surveys, we lack basic information on these species' current distribution and habitat associations. Developments in camera trapping technology allow us to monitor small mammal populations with significantly less time and resources. The Adapted Hunt Drift Fence Technique (AHDriFT) uses a drift fence with a modified camera trap to target small animals. The objectives of my first chapter were 1) to evaluate AHDriFT as a survey method for six state-listed small mammal species in high-elevation forests of South Carolina: the Carolina Red-backed Vole (*Clethrionomys gapperi carolinensis*), Woodland Jumping Mouse (*Napaeozapus insignis*), Eastern Woodrat (*Neotoma floridana*), Masked Shrew (*Sorex cinereus*), Pygmy Shrew (*Sorex hoyi*), and Eastern Spotted Skunk (*Spilogale putorius*); and 2) to identify habitat factors associated with occupancy of these species.

We deployed 60 AHDriFT systems from Summer 2024 to Summer 2026 and collected vegetation and landscape data at each site. We detected all six target species and had sufficient data to model occupancy for Eastern Woodrat, Masked Shrew, Pygmy Shrew, and Spotted Skunk. Spotted Skunk and Eastern Woodrat detection increased with temperature. Pygmy Shrew detection probability was constant, whereas Masked Shrew detection decreased when cameras were knocked over. Spotted Skunk occupancy decreased with distance to water. Eastern Woodrat occupancy increased with deciduous forests, whereas Pygmy Shrew occupancy increased with coniferous forests and Deermouse activity. Masked Shrew occupancy probability was constant. Our findings demonstrate that AHDriFT is an effective method for monitoring sensitive small mammals, confirm these populations have persisted, and identify habitat associations to inform conservation in South Carolina.

A management application of this information is to use small mammals as sentinels for other communities to prioritize conservation and protect co-occurring species. Small mammals have short generation times, high diversity, and strong trophic links, characteristics that could make them a sentinel for biodiversity in the southern Blue Ridge. The objectives of my second chapter were 1) to evaluate whether small mammal diversity indicates diversity of other trophic levels, 2) to evaluate whether small mammal relative activity indicates relative activity of other trophic levels, and 3) to evaluate how microhabitat conditions influence small mammal and shrew relative activity.

We sampled the same 60 AHDriFT sites during the same timeframe. We sampled invertebrates, herpetofauna, and small predators using AHDriFT observations, supplemented with coverboards for herpetofauna, an additional camera per site for small predators, and Berlese-Tullgren funnels for invertebrates. We calculated relative activity, species richness, and Shannon's diversity and investigated which site characteristics best explained these communities. Although diversity and relative activity were mostly explained by habitat, particularly shrub groundcover and distance to water, small predator diversity was best explained by both small mammal diversity and distance to water. We used N-mixture models to evaluate which microhabitat conditions explained small mammal and shrew relative activity. We did not identify a strong explanatory model for shrews. Overall small mammal relative activity was best explained by invertebrate availability, although this relationship was not significant. Collectively, these results suggest this system is complex and requires multi-taxa monitoring rather than focusing on small mammals as a sentinel community.

Significant deviations: None.

Estimated Federal Cost: See final 425.

Recommendations: Close the grant.

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Submitted By: Anna H. Smith, SWAP Coordinator

HABITAT ASSOCIATIONS OF SMALL MAMMALS AND THEIR ROLE AS SENTINELS FOR BIODIVERSITY IN THE SOUTHERN BLUE RIDGE

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In partial Fulfillment
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Master of Science
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By
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PLAIN LANGUAGE ABSTRACT

Small mammals play important roles in healthy forests, but we still know very little about many of the rare and state-listed species living in the Blue Ridge Mountains of South Carolina. Traditional survey methods are expensive and time-consuming, making it difficult to learn where these animals live and what habitats they need. In this study, we tested a new camera-based survey method called the Adapted Hunt Drift Fence Technique (AHDriFT), which is designed to detect small animals more efficiently.

From 2024 to 2026, we set up 60 survey sites in high-elevation forests across the region and recorded habitat conditions at each site. The new method detected all six state-listed small mammal species we were looking for, including some that had not been recorded in South Carolina for about 40 years. We found that eastern woodrats were more likely to occur in deciduous forests, pygmy shrews were more likely where deermice were more active, and eastern spotted skunks were more likely to occur near water. We did not find a habitat feature that explained masked shrews. These results show that AHDriFT is an effective way to monitor rare small mammals and can provide the information needed to guide their conservation.

We also asked whether small mammals could serve as indicators of the overall health of the forest. If they reflected the condition of other groups of animals, managers could use them to monitor biodiversity more efficiently. To test this idea, we measured the diversity and activity of invertebrates, reptiles and amphibians, and small predators at

the same 60 sites. We used the AHDriFT systems along with additional cameras, coverboards, and insect sampling to survey these groups.

We found that the diversity and activity of most animal groups were influenced more by habitat than by the small mammal community. Shrub cover and distance to water were especially important. Only the diversity of small predators was closely related to both small mammal diversity and distance to water. We also examined which habitat features affected the activity of small mammals and shrews. We found a weak relationship between small mammals and insect availability.

Overall, this study shows that AHDriFT is an effective and practical tool for monitoring rare small mammals in South Carolina and provides new information about where these species occur and the habitats they use. However, our results also suggest that no single group of animals can reliably represent the health of the entire forest ecosystem. Monitoring multiple groups of wildlife is likely to provide a more complete picture for conservation and management decisions.

ABSTRACT

There are approximately 40 small mammals in the Blue Ridge ecoregion of South Carolina excluding bats, and of these, 11 are priority species for conservation. Due to the high cost and labor investment of traditional small mammal surveys, we lack basic information on these species' current distribution and habitat associations. Developments in camera trapping technology allow us to monitor small mammal populations with significantly less time and resources. The Adapted Hunt Drift Fence Technique (AHDriFT) uses a drift fence in combination with a modified camera trap to specifically target small animals. The objectives of my first chapter were 1) to evaluate AHDriFT as a viable survey method for six state-listed small mammal species in high elevation forests of South Carolina: the Carolina red-backed vole (*Clethrionomys gapperi carolinensis*), woodland jumping mouse (*Napaeozapus insignis*), eastern woodrat (*Neotoma floridana*), masked shrew (*Sorex cinereus*), pygmy shrew (*Sorex hoyi*), and eastern spotted skunk (*Spilogale putorius*) and 2) to identify habitat factors associated with species occupancy of these six state-listed small mammals.

We deployed 60 AHDriFT systems from summer 2024 to summer of 2026 and collected vegetation and landscape data at each site upon setup. We detected all six target species, some of which hadn't been detected in the state in 40 years. We had sufficient data to model occupancy for eastern woodrat, masked shrew, pygmy shrew, and spotted skunk. Spotted skunk and eastern woodrat detection increased with temperature. Pygmy shrew detection probability was constant, and masked shrew detection decreased when

the AHDriFT cameras were knocked over. Spotted skunk occupancy decreased with distance to water. Eastern woodrat occupancy increased with deciduous forests, whereas pygmy shrew occupancy increased with coniferous forests and deermouse activity. Masked shrew occupancy probability was constant. Our findings demonstrate that AHDriFT is an effective method for monitoring sensitive small mammals and proved their populations have persisted. Furthermore, we collected enough repeated observations to support occupancy modeling, allowing us to identify habitat associations that can inform conservation and management in South Carolina.

A management application to leverage this new information on small mammals for conservation is to use the taxa as a sentinel for other communities in the ecosystem. Managers use sentinels as indicators of ecosystem conditions to prioritize conservation efforts and protect co-occurring species. Small mammals have short generation times, high diversity, and strong trophic links to other taxa. These are characteristics that could make the small mammal community a sentinel for biodiversity in the southern Blue Ridge. The objectives of my second chapter were 1) to evaluate the extent small mammal diversity is a good indicator of the diversity of other trophic levels, 2) to evaluate the extent small mammal relative activity is a good indicator of the relative activity of other trophic levels, and 3) to evaluate how microhabitat conditions important to the southern Blue Ridge influence overall small mammal and shrew relative activity.

To sample other trophic levels, we used the same 60 AHDriFT sites and timeframe as chapter one. We sampled invertebrates, herpetofauna, and small predators for evaluation with small mammals. We used the observations from the AHDriFT

systems to sample herpetofauna and small predators, and took supplemental samples of herpetofauna using coverboards and of small predators using an additional camera at each site. We sampled the invertebrate community at each site using Berlese-Tullgren funnels. We calculated relative activity, species richness, and Shannon's diversity for invertebrates, herpetofauna, and small predators. We investigated which site characteristics, including the small mammal community and other habitat factors, best explained the communities. While communities' diversity and relative activity were mostly explained by habitat, specifically the percent of groundcover by shrubs and the distance to water, small predator diversity was best explained by both small mammal diversity and distance to water. We used N-mixture models to evaluate what microhabitat conditions explain small mammal and shrew relative activity. We did not identify a strong explanatory model for shrews. Small mammal relative activity overall was best explained by invertebrate availability, but these results were not significant. These results collectively suggest this system is complex and requires multi-taxa monitoring rather than focusing on small mammals as a sentinel community.

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INTRODUCTION

SMALL MAMMALS IN THE BLUE RIDGE MOUNTAINS

The Blue Ridge, a temperate mountainous ecoregion in the southeast United States in between the Piedmont and the Appalachian Plateau, is the southernmost range of the Appalachian Mountains. Beginning formation a billion years ago, it is one of the oldest mountain ranges in the world. This long evolutionary history and complex topography have given the flora and fauna of the region ample time for speciation (Newfont, 2012). Today the Southeast is one of the most biodiverse parts of the country with one of the highest rates of endemism across all taxa, and the Blue Ridge in is especially high in diversity of plants and animals (Jenkins et al., 2015).

The high biodiversity and endemism of the Blue Ridge Mountains, coupled with the low rate of protection due to much of the land being privately owned, has led to the area being classified as the highest priority conservation hotspot in the United States (Jenkins et al., 2015). While the southern Blue Ridge area has been occupied for centuries by indigenous people including the Catawba, Eastern Cherokee, Yuchi, and Miccosukee (Native Land, 2024), European settlement and development began in the late seventeenth century. Ever since, trade and industrialization have rapidly increased and transformed the Blue Ridge, to the point where that rich biodiversity is now threatened (Newfont, 2012). To this day much of the land is rapidly being converted for agriculture, timber harvest, and urban development (Surasinghe and Baldwin, 2014). Furthermore, climate change data show the area is a key conservation concern for climate resiliency (Lacher et al., 2019). Many high elevation, mountainous sites are especially vulnerable as the steep topography creates temperature gradients that produce distinct

climate zones in shorter distances than latitude changes would suggest. These concentrated, stratified climate zones run out of land to move uphill as the ecosystem warms and can rapidly disappear (Diaz et al., 2003), leaving climate sensitive species stranded with no suitable habitat. Thus, the compounding effects of land use change and climate change necessitate careful management to conserve this biologically rich region.

Terrestrial small mammals, defined here as non-flying mammals with a mean adult body size of 5 kilograms or less, represent a diverse and ecologically significant category of wildlife in the Blue Ridge Mountains. In the Blue Ridge Mountains, specifically the Great Smoky Mountains National Park, there are at least 68 mammal species (Linzey, 2016). Of those approximately two thirds are small mammals, which can be found across different terrestrial habitat types, including meadows and woodlands. Small mammals occupy low trophic levels and are predated upon by a wide variety of birds, reptiles, and other mammals, often functioning as a primary food source for those animals (Pivorun et al., 2006). Many species of small mammals are granivores and play a critical role in seed dispersal, which drives the successional structure of the plant communities (Pauli et al., 2005). Insectivorous small mammals regulate invertebrate populations (Linzey, 2016). However, most small mammal species in the Great Smoky Mountains have declined in the last 50 years, mostly due to deforestation (Linzey, 2016). Since the glacial retreat following the Pleistocene, several species (e.g., water shrew, northern flying squirrel, red squirrel, and rock vole) typically found more northward have refugial, or disjunct, populations in the high elevation zones of the Blue Ridge, which are now threatened by climate change (Runck and Cook, 2005; Linzey, 2016). As the southernmost range of the Appalachian Mountains, the Blue Ridge represents the edge of the ranges for many of these high elevation species, which puts them at risk of extirpation as climate regimes shift uphill and northward.

The South Carolina State Wildlife Action Plan (SWAP) lists 11 small mammals as species of management interest due to their decline or risk of extirpation from the state in the Blue Ridge ecoregion. In particular, at least six of the SWAP-listed species are present in high-elevation forest located in the northwest, bordering Georgia and North Carolina: these include the Carolina red-backed vole (*Clethrionomys gapperi carolinensis*), woodland jumping mouse (*Napaeozapus insignis*), eastern woodrat (*Neotoma floridana haematoreia*), masked shrew (*Sorex cinereus*), pygmy shrew (*Sorex hoyi*), and eastern spotted skunk (*Spilogale putorius*). Despite high conservation and management interest in these species, they are historically understudied due to the special monitoring challenges this group of animals presents. Monitoring efforts are often labor intensive and expensive, involving live trapping and pitfalls traps with or without drift fences. Sherman traps on a grid are a common method used (Loeb, 1999; Kaminski et al., 2007; Conrad and Reitsma, 2015), but must be checked every day to ensure the animal survives capture. This can also be an issue with another common method, drift fences with pitfalls (Brannon, 2005). In addition to challenges with time and labor, trapping with Shermans or pitfalls can also lead to injury or even mortality of the study species, which is both an ethical concern and counterproductive when surveying sensitive or management interest species (Martin et al., 2017). This is especially an issue when pitfalls are not checked every day (Laerm et al., 1995).

Game cameras have become a popular method to detect mammals as costs have dropped with improving technology, but even though this method saves time and resources, it is difficult to use them to detect small mammals (Martin et al., 2017; Amber et al., 2021b; White et al., 2023). A novel survey method combining drift fences with camera traps to combat these limitations has been employed successfully for small mammal and herpetofauna studies in the

Midwest (Amber et al., 2021a; Amber et al., 2021b; White et al., 2023) and Florida (Martin et al., 2017). The Adapted-Hunt Drift Fence Technique (AHDriFT) uses a drift fence that leads on either side to an overturned bucket fixed with a camera to the top. A small opening is cut out of the side of the bucket so the animal may follow the drift fence, enter the overturned bucket, and have its image captured. This technique is less invasive and more cost effective to the researchers trying to detect small mammals (Martin et al., 2017, Amber et al., 2021b), but this technique had not been used to monitor at-risk small mammals in the Blue Ridge Mountains.

In this study I applied the AHDriFT method to evaluate the community ecology of small mammals in the Blue Ridge Mountains of South Carolina. The goal of the study was to better understand the distribution and ecology of at-risk small mammals in this region. In Chapter 1, I surveyed areas of past record of the SWAP listed species to test how well the AHDriFT system works to monitor small mammals in the South Carolina Blue Ridge Mountains and evaluated support for hypothesized habitat features associated with their occurrence. In Chapter 2, I examined relationships between small mammal diversity and relative activity with the diversity and relative activity of other trophic levels to test the hypothesis that small mammals are good sentinel species for biodiversity in this ecosystem. Collectively, my project provides a better understanding of how to monitor for at-risk small mammals and the ecology and importance of small mammals in the southern Blue Ridge Mountain ecosystem.

FOCAL SPECIES NATURAL HISTORY

Carolina Red-Backed Vole

The Carolina red-backed vole (Clethrionomys gapperi carolinensis) is limited to high elevation sky islands above 610 meters to 915 meters in the southern Blue Ridge. It uses mesic mixed forests, mesic deciduous hardwood forests, and high elevation coniferous forests with

beech gaps. Common characteristics of its habitat include coarse woody debris, rotting logs, moss covered rocks, exposed crevices, mountain balds, Rhododendron thickets, and bogs (Menzel et al., 1999; South Carolina Department of Natural Resources, 2020). Its diet includes seeds and nuts, flowers, berries and fruits, fungi, roots, bark, ferns, lichens, invertebrates, and green vegetation. It is predated upon by bobcats, weasels, foxes, snakes, and birds of prey. It builds globular nests of plant and moss under rocks, roots, logs, or stumps (Pivorun et al., 2006; Linzey, 2016).

In South Carolina the Carolina red-backed vole was historically found in Oconee and Pickens County, and most recently sighted in 1994 near Walhalla Fish Hatchery and Sassafras Mountain (NatureServe, 2024a). Its conservation status in South Carolina is S2S3, imperiled/vulnerable in the state (South Carolina Department of Natural Resources, 2025). The subspecies -*carolinensis* found in the state ranges from Virginia and West Virginia, though the exact line requires more genetic analysis (South Carolina Department of Natural Resources, 2020). Primary threats to its status are thought to be habitat loss due to land development and a fungal disease hemlock woolly adelgid (*Adelges tsugae*) that affects its habitat, (NatureServe, 2024a) (Supplemental Material Table A1.1).

Woodland Jumping Mouse

The woodland jumping mouse (*Napaeozapus insignis*) is limited to cool, moist habitat at high elevations exceeding 1400 ft. It uses hemlock stands and sheltered cove hardwood stands. Common habitat characteristics include herbaceous cover, Rhododendron-dominated stream sides, and the presence of *Endogone* and *Glomus* fungi (South Carolina Department of Natural Resources, 2020). Its diet includes seeds, roots, fruits, insects, other invertebrates, and fungi, especially *Endogone.* It is predated upon by weasels, skunks, minks, bobcats, snakes, and birds

of prey. It nests in brush piles, under fallen logs, or underground. It typically hibernates from October to April (Pivorun et al., 2006).

There is very little information about woodland jumping mouse population densities and locations in South Carolina, but they are only found at high elevation sites (South Carolina Department of Natural Resources, 2020) and may prefer north facing slopes (Brannon, 2005). Historic records occur in Greenville and Pickens counties, where the most recent detections were in 1980 (NatureServe, 2024b). Its conservation status in South Carolina is S2S3, imperiled/vulnerable in the state (South Carolina Department of Natural Resources, 2025). The subspecies for the area is -*roanensis*, all other subspecies are out of range (Whitaker & Wrigley, 1972). Primary threats to its status are thought to be habitat loss due to land development, *Adelges tsugae* infection in its habitat, and climate change shifting its distribution northward (Cassola, 2016a; NatureServe, 2024b) (Supplemental Material Table A1.1).

Eastern Woodrat

The eastern woodrat (*Neotoma floridana haematoreia*) is a generalist found across many habitat types including dry or mesic deciduous forests, mixed deciduous forests, coves, bottomlands, swamps, pine heath or early successional stands, rock outcrops, boulder fields, and cliffs (South Carolina Department of Natural Resource, 2020). It is generally an opportunistic forager, with a diet that includes seeds and nuts, berries and fruits, leaves, stems, tubers, mushrooms, and fungi (Pivorun et al., 2006, McMurry et al., 1993). It is predated upon by skunks, weasels, foxes, snakes, and birds of prey (Pivorun et al., 2006). It gets the nickname "packrat" from its nesting habits, where it builds nests underground, in hollow logs, under stumps, in trees, in stream banks, or in old buildings. The nests range in size from the size of a football, to 3 ft in diameter piles made of food, sticks, bones, bark, feathers, duns, paper, colored

glass, rags, and other human trash (Pivorun et al., 2006). They are also known to use windthrown root masses to nest in (Knowles and Burger, 2008) or rock crevices with fissures (Rossell et al., 2008).

Eastern woodrats are found in a variety of habitats across South Carolina, though the distribution in the Midlands and Upper Piedmont are not well understood. They also utilize the Coastal Plains and Sandhill (South Carolina Department of Natural Resources, 2020). The most recent detections in the Blue Ridge were in Greenville and Pickens Counties in 2005 (NatureServe, 2024c). Its conservation status in South Carolina is S3S4, vulnerable/apparently secure in the state (South Carolina Department of Natural Resources, 2025). There formerly were two subspecies in the state, -*haematoreia* for the mountain populations and -*floridana* for the plains, but genetic analysis showed them to be the same, and all the South Carolinian specimen are now recognized as the subspecies -*haematoreia* (South Carolina Department of Natural Resources, 2020). Primary threats to its status are thought to be habitat loss due to land development and increased density of predator populations (Cassola, 2016b; NatureServe, 2024c) (Supplemental Material Table A1.1).

Masked Shrew

The masked shrew (*Sorex cinereus*) is limited to elevations over 610 meters (Linzey, 2016). It uses mesic mixed forests, mesic deciduous hardwood forests, dry deciduous forests, and eastern hemlock ravine forests with thick understories, high soil moisture, high organic matter, logs, stumps, rocks, and dense leaf litter (South Carolina Department of Natural Resources, 2020). It may prefer north facing slopes (Laerm et al., 1995). Its diet includes terrestrial snails, slugs, insects, ants, spiders, worms, dead vertebrates, and occasionally plants (Pivorun et al., 2006). It can be predated upon by domestic cats and dogs (NatureServe, 2024d), but its wild

predators are foxes, weasels, bobcats, snakes, and birds of prey. It builds nests 3-4” in diameter out of dried leaves and grass under logs, roots, or rocks (Pivorun et al., 2006).

There is little information about the distribution of masked shrews in South Carolina, but they have only been found in high elevation sites (South Carolina Department of Natural Resources, 2020). Historic detections occurred in Oconee County (NatureServe, 2024d) with the most recent detection being near the Walhalla Fish Hatchery in 1994 (Laerm et al., 1995). Its conservation status in South Carolina is S2, imperiled in the state (South Carolina Department of Natural Resources, 2025). The subspecies for the area is -*cinereus*, all other subspecies are out of range (NatureServe, 2024d). Primary threats to its status include habitat loss due to land development, predation by domestic cats and dogs, litter in their environment, and the *Adeleges tsugae* fungal infection compromising habitat (NatureServe, 2024d) (Supplemental Material Table A1.1).

Pygmy Shrew

The pygmy shrew (*Sorex hoyi*) is the smallest mammal in the United States and is rarely observed (Linzey, 2016). It uses mesic mixed forests, mesic deciduous hardwood, dry deciduous forests, and eastern hemlock ravine forests (South Carolina Department of Natural Resources, 2020). Habitat characteristics it prefers include deep leaf litter and rotting logs, but it can be found on ridge tops and mountains or in grassland and forest edges (Pivorun et al., 2006). Its diet includes insects, spiders, ants, seeds, berries, and worms. It is predated upon by foxes, snakes, and birds of prey. It nests just below the ground surface under logs or roots or in burrows (Pivorun et al., 2006; Linzey, 2016).

There is little information of population distribution and densities of the southern pygmy shrew in South Carolina (South Carolina Department of Natural Resources, 2020). There are

some historic records in Pickens County (NatureServe, 2024e) near the Walhalla Fish Hatchery (Laerm et al., 1995). The most recent detection was in 2000 in Pickens County (NatureServe, 2024e). Its conservation status in South Carolina is S2, imperiled in the state (South Carolina Department of Natural Resources, 2025). The subspecies for the area is -*winnemana*, all other subspecies are out of range (NatureServe, 2024e). Primary threats to its status include habitat loss due to land development, predation by domestic cats and dogs, and litter in their environment (NatureServe, 2024e) (Supplemental Material Table A1.1).

Eastern Spotted Skunk

The Alleghanian spotted skunk (*Spilogale putorius*) (also called eastern spotted skunk) is a mesocarnivore known for its use of a musk (methyl mercaptan) defensively sprayed from its anal glands (Pivorun et al., 2006). It uses woods and brush, and can be found near farmyards and old fields. Preferred habitat characteristics include dense cover, coarse woody debris, rock outcrops, and barns (South Carolina Department of Natural Resources, 2020). It is somewhat a habitat generalist, but shows a preference for areas with complex structure including understory vegetation and coarse woody debris, such as drainages (Eng & Jachowski, 2019a; Harris et al., 2020; Thorne et al., 2017). Its diet includes beetles and other insects, rodents and rabbits, worms, bird eggs, and occasionally fruit. It is predated upon by great horned owls, bobcats, and domestic cats and dogs (Pivorun et al., 2006; Linzey, 2016). It dens in underground burrows located beneath farm buildings or in hollow logs, rock crevices, or lumber slab piles (Linzey, 2016). It may also use mammal or gopher tortoise burrows (Harris et al., 2020; Sprayberry & Edelman, 2018). They normally den alone, but they will share dens, especially in winter (Harris et al., 2020; Lesmeister et al., 2008).

This species of spotted skunk ranges throughout the Appalachian mountains and down the southeastern coast of the United States (McDonough et al., 2022). In South Carolina the Alleghenian spotted skunk has historically occurred in Aiken, Edgefield, Greenville, Oconee, and Pickens County. It was last sighted in 2021 in Greenville and Pickens Counties (NatureServe, 2024f) Its conservation status in South Carolina is S2, imperiled in the state (South Carolina Department of Natural Resources, 2025). Primary threats to its status include overharvest by fur trappers, pesticide use, habitat fragmentation and destruction, altered predator communities, and disease (Eng & Jachowski, 2019b; Gompper & Hackett, 2005; Gompper & Jachowski, 2015; NatureServe, 2024f; Sprayberry & Edelman, 2018; Thorne et al., 2017) (Supplemental Material Table A1.1).

STUDY AREA DESCRIPTION

This study occurred in the Blue Ridge Mountain Range in South Carolina, in the northwest corner of the state, bordering Georgia and North Carolina. The lower elevation threshold for the woodland jumping mouse is 426 meters (1400 feet), so that threshold was used as the cutoff for this study (see study area section of Chapter 1 for rationale). The highest elevation found in the area is 1018 meters (3340 feet). The mean elevation is 737 meters (2418 feet), with a range of 590 meters (1936 feet).

The study was also limited to areas I could access with permits, meaning I used publicly owned land. The primary landowners throughout the study area were the US Forest Service and the South Carolina Department of Natural Resources, but there are also some areas owned by state parks. There are a variety (30 unique types identified) of deciduous, coniferous, and mixed forest types in the area (Table 1.1). The most common forest type found was white pine at 2282 total hectares or 14.6% of the study area. The next most common forest types in order of total

hectares or percentage of the study area are White oak-northern red oak-hickory, Shortleaf pine-oak, White pine-cove hardwood, Chestnut oak-scarlet oak-yellow pine, and White pine-upland hardwood. All other forest types each compose less than 5% of the total study area, though combined they make up about 40% of the total study area. More detail on the specific selection of monitoring sites within this region can be found in the Methods section of Chapter 1.

CHAPTER ONE: HABITAT ASSOCIATIONS OF AT-RISK SMALL MAMMALS IN THE BLUE RIDGE MOUNTAINS OF SOUTH CAROLINA

INTRODUCTION

Small mammals are a diverse and ecologically important group that occupy low trophic levels and serve as a primary food source for many birds, reptiles, and mammals (Pivorun et al., 2006). They also play key functional roles in ecosystems, granivorous and mycophagous species influence plant community succession through seed and spore dispersal (Pauli et al., 2006; Stephens et al., 2021), while insectivores help regulate invertebrate populations (Linzey, 2016). However, many small mammal populations are declining due to increased predation pressure from both wildlife and domestic animals, habitat loss and fragmentation, and climate change (Alexiou et al., 2025; Medd et al., 2025; Simelane et al., 2018).

In temperate ecosystems of the eastern United States, warming temperatures and declining habitat availability pose particularly acute threats to high-elevation species with disjunct southern populations. Following glacial retreat after the Pleistocene, several species today more commonly associated with northern latitudes [e.g., water shrew (*Sorex palustris*), northern flying squirrel (*Glaucomys sabrinus*), red squirrel (*Sciurus vulgaris*), and rock vole (*Microtus chrotorrhinus*)] persist in refugial populations in the southern Appalachian highlands, where they are now highly vulnerable to climate warming (Runck and Cook, 2005; Linzey, 2016). As the southernmost extent of the Appalachian Mountains, the Blue Ridge marks the range edge for many of these species, placing them at heightened risk of extirpation as suitable climate conditions shift upslope and northward (Diaz et al., 2003).

In South Carolina, the 2015 South Carolina State Wildlife Action Plan (SWAP) lists 11 small mammals as species of management interest due to their decline and risk of extirpation from the state (South Carolina Department of Natural Resources, 2020). In particular, at least six of the SWAP-listed species are present in the Blue Ridge located in the northwest corner of the state, bordering Georgia and North Carolina: these include the Carolina red-backed vole (*Clethrionomys gapperi carolinensis*), woodland jumping mouse (*Napaeozapus insignis*), eastern woodrat (*Neotoma floridana haematoreia*), masked shrew (*Sorex cinereus*), pygmy shrew (*Sorex hoyi*), and eastern spotted skunk (*Spilogale putorius*). The exact distribution and habitat use of many small mammal populations are not well understood in South Carolina, and the last observations from state survey records for some species ranges from 2021 for the eastern spotted skunk back to 1980 for the woodland jumping mouse; the woodland jumping mouse and the Carolina red-backed vole (last observed in 1994) have only two recorded observations each in the state database (South Carolina Department of Natural Resources, 2020). To inform their conservation status and guide future restoration efforts, there is an urgent need to clarify these species' status, ranges, and habitat associations.

Small mammal surveys typically use Sherman and pitfall traps, both of which capture animals in either metal boxes or pits. These are labor-intensive and tend to be most effective when used together to maximize detection (Garden et al., 2007). The Adapted Hunt Drift Fence Technique (AHDriFT) was developed to reduce mortality, cost, and field effort (Martin et al., 2017; White et al., 2023) and has shown strong potential for monitoring small mammals and herpetofauna (Amber et al., 2021a; Martin et al., 2017; Amber et al., 2021b; White et al., 2023). The AHDriFT is a drift fence that terminates in either end with a camera trap modified to target small animals. In northeastern Indiana, AHDriFT produced 23× higher Shannon diversity and

1.5–5× more detections per dollar in 90% less time than Sherman traps (White et al., 2023). In Florida, it reduced field time by 95% with no wildlife mortality (Martin et al., 2017), and in Ohio it detected four mammal species missed by traditional methods, though performance was lower for rare or low-mobility species (Amber et al., 2021b). To date, there have been no attempts to deploy AHDriFT systems in the southern Blue Ridge region to monitor for small mammals.

My first objective was to evaluate whether AHDriFT could detect six state-priority small mammal species in high-elevation forests of South Carolina. I deployed systems across Blue Ridge forest types (Table 1.1) based on historical records and published habitat associations for each focal species (Supplemental Material Table A1.1). My second objective was to identify habitat factors associated with species occupancy. I hypothesized that (1) red-backed voles, masked shrews, and woodland jumping mice would be associated with food and water availability (Brannon, 2005; Laerm et al., 1995; Pauli et al., 2006); (2) structural cover such as shrubs and coarse woody debris would promote occurrence by providing thermoregulation, refuge, and nesting sites for all target species (Brower & Cade, 1966; Loeb, 1999; Sprayberry & Edelman, 2018); and (3) interspecific competition with generalist rodents would influence occupancy (Boonstra & Krebs, 2012; Conrod & Reitsma, 2015).

METHODS

Study Area

This study occurred in the Blue Ridge Mountain Range in South Carolina, in the northwest corner of the state, bordering Georgia and North Carolina, in Oconee and Pickens counties. I focused on high elevation Appalachian oak forests, and set the low elevation threshold to 426 meters (1400 feet) based on the lower elevation limit of woodland jumping mice in South Carolina

(South Carolina Department of Natural Resources, 2020). Of the three focal species thought to occur only in high elevation forests in South Carolina (the Carolina red-backed vole, masked shrew, and woodland jumping mouse), the woodland jumping mouse has the lowest elevation threshold, which makes this an ideal threshold to study the high-elevation communities that are vulnerable to climate change (Diaz et al., 2003). The highest elevation found in the area is 1018 meters (3340 feet). The mean elevation is 737 meters (2418 feet), with a range of 590 meters (1936 feet).

There are 30 unique types of deciduous, coniferous, and mixed forest types in the area (Francis Marion and Sumter National Forest FSVeg Spatial; Table 1.1). The most common forest type found was white pine at 2282 total hectares or 14.6% of the study area. The next most common forest types in order of total hectares or percentage of the study area are White oak-northern red oak-hickory, Shortleaf pine-oak, White pine-cove hardwood, Chestnut oak-scarlet oak-yellow pine, and White pine-upland hardwood. All other forest types each compose less than 5% of the total study area, though combined they make up about 40% of the total study area.

Site Selection

I selected study sites for AHDriFT placement using three criteria. First, to ensure access, I selected only publicly owned federal and state lands, including state parks, US Forest Service lands, and South Carolina Department of Natural Resources lands. Second, sites were required to have reasonable access by buffering roads and trails by 500 m, while excluding areas within 200 m of roads to minimize traffic disturbance. Third, I applied habitat criteria to increase detection probability of the focal small mammals. All focal species show preferences for coarse woody debris and understory vegetation, which are most abundant in drainages (Bowman et al., 2001; Brannon, 2005; Ford & Rodrigue, 2001; Gooley & Schauber, 2018; Loeb, 1999; Menzel et al.,

1999; Pauli et al., 2006). Accordingly, I prioritized sampling in drainages by buffering streams by 50 m, reflecting documented preferences of red-backed voles, masked shrews, woodland jumping mice, and eastern spotted skunks (Brannon, 2005; Brower & Cade, 1966; Campbell et al., 2010; Ford & Rodrigue, 2001; Kirkland & Griffin, 1974; Laerm et al., 1995; Linzey, 2016; Orrock et al., 2003; Pauli et al., 2006; Pivorun et al., 2006). In addition to drainages, I also prioritized ridgetops (areas above 762 m (2500 ft)), because those often xeric, coniferous habitats have been known to have been used by pygmy shrews and red-backed voles (South Carolina Department of Natural Resources, 2020).

To ensure I sampled across the full spatial and elevational range of my 6 target species, I stratified the portion of the study area meeting my three criteria into low (426–610 m), medium (610–762 m), and high (>762 m) elevation classes. Low and medium elevations were within the drainage buffer, while high elevations occurred on ridgetops. This approach resulted in approximately one-third of my sites being located on ridgetops and steep slopes, the relatively less abundant habitat type in my study system. For each of these elevation categories I randomly generated 30 points, buffered by 805 meters (0.5 miles) to reduce spatial autocorrelation (the exception being around the Ellicott Wilderness Area, some points were spaced 400–480 meters apart to increase sampling intensity due to a high number of historical detections). My final site selection criteria was to prioritize the 20 sites that most overlapped with clusters of historical detections in each of my 3 elevational strata, for a total of 60 study sites.

AHDriFT System

I placed an AHDriFT system at the nearest accessible point within 300 meters to the centroid of each study site, selecting the exact deployment location based on the availability of understory cover and CWD, but also space to lay out the 10-m long drift fence. Following the

design of Martin et al. (2017), I installed a 10-m long by 1-m tall silt fence supported by 5 evenly spaced stakes. I brushed away litter and duff along the fence line but replaced it after staking in the drift fence so that the fence is partially buried, using large sticks and logs to anchor the fence down. At each end I placed a 19-liter overturned bucket with a 6-cm by 6-cm box-shaped hole carved into opposite sides of the bucket, ensuring the holes were placed parallel to the drift fence so that the animals were guided by the drift fence into the hole. I placed 2.5-cm by 10-cm boards at an angle towards the inside of the bucket to further funnel the animal into the bucket. On the underside of the bucket, I bolted a Browning TrophyCam HD trail camera facing the ground, so that the animals entering the bucket had their photo taken from above.

I modified the cameras by attaching a 4.00 diopter reading glass lens to help the camera focus the correct distance (30 cm) to the ground (Meek and Cook 2022), using caulk around the lens to avoid condensation. After trial and error, I also placed 20 layers of masking tape over the infrared flash to reduce photograph washout. I set the camera to take 3 image-burst photos when triggered, with a 3 second delay between trigger events. To monitor larger animals that visit the fence but did not go in the buckets, I placed a third camera on a nearby tree facing the system.

The system once placed was left out for one year. I performed a first check when the system had been out for one week to make sure it was operating correctly and then checked the system once a month to switch out the SD cards and replace the batteries. After the first year, the systems were moved to 30 new sites and monitored for another year with monthly checks.

Site-level Measurements

To evaluate support for each of my hypotheses regarding habitat factors associated with my focal species, at each site I recorded coarse woody debris, understory vegetation cover, canopy cover, understory and overstory vegetation type, leaf litter depth, and soil temperature at

each site. To measure coarse woody debris I used line transect sampling (Behjou & Mollabashi, 2013), where I walked a 10-m transect perpendicular off either side of the midpoint of the drift fence. Along the transect I counted logs or branches >1 m in length and classified them by length class (<2 m, 2-4 m, >4 m), diameter (<20 cm, 20-30 cm, >30 cm at midpoint), stage of decomposition per the British Columbia Wildfire Service categories (Ministry of Forests, Lands and Natural Resource Operations & Forest Analysis and Inventory Branch, 2018), and whether a rootmass was present. These transect data were summed to give each site a score for the volume of coarse woody debris present.

I recorded canopy cover with a densiometer at the midpoint and both ends of the AHDriFT system, and averaged the three values. I estimated to the nearest 10% the amount of deciduous versus coniferous coverage in the canopy. To determine understory vegetation cover, I estimated the percentage to the nearest 10% of ground cover by shrubs in the immediate (10 m radius) area. I used a hydrology layer (*The National Map*, n.d.) in ArcGIS to calculate the distance of the site from water, and a DEM layer (WorldElevation/Terrain (ImageServer), n.d.) to calculate elevation. To estimate litter depth, I used a trowel with centimeter markings on it, pushed it into the leaf litter until I hit soil, and then recorded the depth in centimeters. I repeated this three times per site, at the midpoint of the drift fence and by either bucket and averaged the values.

Data Processing

The photos collected from the cameras were processed by myself and trained technicians using Wildlife Insights (app.wildlifeinsights.org) and Timelapse2 software (*Timelapse – from Camera Trap Images to Data*, n.d.). Wildlife Insights uses artificial intelligence (AI) to identify wildlife in photos down to species, but the model is not well-trained for AHDriFT oriented

photographs. Therefore, we manually identified species in all photographs, including blanks, using the program Timelapse2. This program allowed me to create a custom template of fields to fill for each photo which included site, camera ID, date, time, species, and whether the bucket was knocked over or not. Technicians and myself could then use the Timelapse2 interface to populate the template for all the photos.

Analysis

I used single-species single-season occupancy modeling in the R package *unmarked* (Fiske & Chandler, 2011) to determine what habitat characteristics from the candidate models best explained occupation of the 6 focal species (MacKenzie et al., 2017). I modeled each of the 6 focal species independently, using data from the AHDriFT cameras and the third outside camera for each site. I set the survey occasion length to 6 weeks, starting at the day each site was set up. I used 5 survey occasions (30 weeks) for each site. I used the two-step, hierarchical process for model fitting (Wharff et al. 2025), first fitting detection models and then fitting occupancy probability models with the top-ranked detection probability model using AIC. I used a goodness-of-fit test from the *AICcmodavg* package in R (Mazerolle, 2023) on the global model with all detection and occupancy covariates to choose between AIC and QAIC for each species with the overdispersion parameter, c-hat (MacKenzie and Bailey 2004).

Detection Models

I evaluated support for 8 a-priori detection probability (*p*) models, including a null model, for each of the 6 focal species (Table 1.2). I developed 5 models with single covariates: average maximum daily temperature (*temp*), number of days the camera was knocked over (*dnf*), slope grade at the site (*slope*), *Peromyscus* relative activity index (*pero*), and season (*season*). I used climate data from the National Weather Service to calculate the average maximum daily

temperature for each site and survey occasion by using the *extract* tool from the R package *terra* (Hijmans, 2025) and *st_nearest_feature* from the R package *sf* (Pebesma, 2018) to find the closest weather tower for each site by geometric distance. The available towers were Caesars Head, Highlands, Jocassee 8 WNW, Lake Toxaway 2 SW, Long Creek, Pickens, Table Rock Reservoir, and Walhalla. I predicted temperature would have a negative association to detectability as game cameras use temperature differentials between wildlife and their environment to trigger infrared sensors (Swann et al., 2004). To calculate the total number of days the camera was knocked over at each site for each survey occasion, I added the number of days the bucket was down for both cameras at each site. This gave a total number between 0 and 84 for the 42 day survey occasion. I predicted the number of days the camera was knocked over would have a negative association with detection for all species, as the buckets were not providing cover for small mammals. I used the *extract* tool from the R package *terra* and a DEM layer (Hijmans, 2025; *WorldElevation/Terrain (ImageServer)*, n.d.) to calculate the slope grade at each site. I predicted slope would have a negative association with detection for each species because on steeper slopes, small mammals might travel downhill rather than exploring inside the camera bucket. For the *Peromyscus* relative activity index, I calculated the number of independent detections of *Peromyscus* that were at least 60 minutes apart (Amber et al., 2021a, 2021b; Martin et al., 2017; White et al., 2023) for each 42 day survey period. *Peromyscus* are territorial and could defend the camera bucket as microhabitat with cover, so I predicted they would have a negative association with detection of all species (Conrod & Reitsma, 2015). I used season as a categorical covariate with the categories Spring, Summer, Autumn, and Winter. I assigned the season for each survey occasion from the month of the midpoint date of the occasion, with March - May as Spring, June - August as Summer, September - November as

Autumn, and December - February as Winter. However, we did not have data for more than 6 occasions, so there were no Spring samples. I predicted detection probability would be highest in Summer, when the entrance to the bucket is less likely to be blocked by falling leaves or snow. I scaled all continuous covariates prior to model fitting. I also developed a sub-global model: *temp* + *dnf*, to look at a combination of temperature and days knocked over. In addition, I evaluated support for null and global models. I ranked models using Akaike's Information Criterion adjusted for small sample size (AICc), and carried over the top-ranked detection probability model for each species to fit occupancy models.

Occupancy Models

I created 11 a-priori occupancy models (ψ) to evaluate support for my hypotheses for each of the 6 focal species (Table 1.3). For my first hypothesis that small mammal occupancy is associated with food and water availability, I evaluated support for a species-specific predicted effect of canopy type (*cft*), and litter depth (*ld*) (Table 1.3). I predicted that red-backed voles would be more common in conifer stands, as they depend on those seeds for food (Franc et al., 2004; Fuller et al., 2004; Linzey, 2016; Pauli et al., 2006; Pivorun et al., 2006). I also predicted that red-backed vole, masked shrew, and woodland jumping mouse would occur more frequently in mesic forest stands near drainages (*dw*) with closed canopy and higher soil moisture (Brannon, 2005; Brower & Cade, 1966; Campbell et al., 2010; Ford & Rodrigue, 2001; Kirkland & Griffin, 1974; Laerm et al., 1995; Linzey, 2016; Orrock et al., 2003; Pauli et al., 2006; Pivorun et al., 2006). For my second hypothesis, that small mammal occupancy is associated with shrubby vegetation and coarse woody debris, I predicted that percent of the ground covered by shrubs (*ps*), and volume of coarse woody debris at the site (*cwd*) would have a positive association with occupancy for all target species (Bowman et al., 2001; Brannon, 2005; Campbell et al., 2010;

Ford et al., 1999; Ford & Rodrigue, 2001; Gooley & Schauber, 2018; Kirkland & Griffin, 1974; Laerm et al., 1995; Linzey, 2016; Loeb, 1999; Orrock et al., 2003; Pauli et al., 2006; Pivorun et al., 2006; Rossell et al., 2009; Table 1.3) I also predicted that because the masked shrew and woodland jumping mouse are adapted to colder climates, they would be more likely to occupy higher elevation, so I evaluated support for an effect of elevation (*elevation*) (Brannon, 2005; Laerm et al., 1995). I tested my covariates for collinearity and found that elevation was positively correlated with *Peromyscus* relative activity and negatively correlated with distance from water, so I removed the elevation model from my candidate set. My third hypothesis was that specialist small mammals would be outcompeted and therefore less likely to occupy sites where there are abundant generalist small mammals. I predicted that increasing activity of *Peromyscus* (*pero*) will decrease the probability of occupancy of red-backed vole, woodland jumping mouse, masked shrew, and pygmy shrew (Boonstra & Krebs, 2012; Conrod & Reitsma, 2015; Orrock et al., 2003). I also evaluated support for 3 multivariate models combining the covariates described above. I predicted cover from both shrubby vegetation and the volume of coarse woody debris would have positive effects on occupancy, so I evaluated a sub global model with both covariates (*ps + cwd*). I predicted a combination of ground cover and foraging availability would have species-specific effects on occupancy, so I fit a sub-global model with litter depth and percent of the ground covered in shrubs (*ld + ps*) (Table 1.3). I also predicted occupancy will be affected by both the moisture and cover covariates, so I fit a sub-global model with distance to water and percent of the ground covered in shrubs (*dw + ps*). I evaluated a null and global model for 11 total models in the candidate set. Prior to analysis, I scaled all the continuous covariates. I used a goodness of fit score on the global models to calculate c-hat. I ranked models based on AIC weight and examined significant effects of covariates within the

top-ranked model based on if the 85% confidence intervals of β estimates overlap zero (Arnold 2010). If c-hat was greater than 1 I used QAIC instead.

RESULTS

Including only AHDriFT cameras I had 530,203 photos, and including the third camera I had 735,002 photos. I identified 6,399 independent detections of animals and identified 41 unique species across 59 sites using the AHDriFT cameras. This included 25 mammal species, 10 reptiles, and 6 amphibians. I also observed two additional genera of mammals, but *Peromyscus* and *Sylvilagus* individuals could not be reliably identified to species. Including the third camera that viewed the system, I observed 10,418 independent detections of 45 species across 59 sites, with 4 more mammal species from the third camera. Across all cameras I had 225 mice, 34 shrews, and 2 salamanders that could not be identified to species (Table 1.4). I had 535 complete unknowns that are not included in either totals for independent detections.

The season with the most detections for each target species varied, but all six species were detected in autumn. Masked shrew and pygmy shrew were detected most frequently in autumn, while eastern woodrat and eastern spotted skunk were detected most frequently in summer. Woodland jumping mouse was detected only once in autumn and Carolina red-backed vole was detected once in autumn and once in winter (Table 1.5). I did not have data for spring.

Eastern Woodrat

I observed 300 independent detections of eastern woodrats (*Neotoma floridana haematoreia*) at 17 independent sites (Figure 1.1). The mean latency to initial detection was 64 days (range 3 – 194). The goodness of fit score (c-hat) for the global detection model was 2.72, so I used QAIC to rank my candidate detection models and inflated my standard error for the 85% confidence interval. There was model selection uncertainty, with 5 models (temperature,

null, number of days the camera was knocked over, temperature + number of days the camera was knocked over and slope) in the 80% confidence set within the candidate models evaluating detection probability. The top-ranking detection model of the candidate set was temperature (31% model weight; 1.12× more support than the next model) (Supplemental Material Table A1.2). The scaled effect size of temperature was 0.65 (85% confidence interval = 0.02 – 1.28), suggesting there was a positive association between eastern woodrat detection and increasing temperature.

The goodness of fit score (c-hat) for the global occupancy model was 2.52, so I used QAIC to rank my candidate models and inflated my standard error for the 85% confidence interval. There was model selection uncertainty, with 5 models (forest type, *Peromyscus* relative activity index, the null, volume of coarse woody debris, and litter depth) in the 80% confidence set. The top-ranked occupancy model was forest type (27% model weight; 1.21× more support than the next model) (Supplemental Material Table A1.3). The scaled effect size of forest type (calculated as percent of conifers in the canopy) was -1.08 (85% confidence interval = -2.13 – -0.03), suggesting a negative association between eastern woodrat occupancy and coniferous forest types.

Eastern Spotted Skunk

I observed 81 independent detections of eastern spotted skunks (*Spilogale putorius*) at 24 independent sites (Figure 1.2). The mean latency to initial detection was 68 days (range 5 – 191). The goodness of fit score for the global detection model was 1.08, so I used AIC to rank my candidate detection models. There was model selection uncertainty, with 2 models (temperature and temperature + number of days the camera was knocked over) in the 80% confidence set. The top-ranked detection model was temperature (67% model weight; 2.58× more support than the

next model) (Supplemental Material Table A1.4). The scaled effect size of temperature was 0.85 (85% confidence interval = 0.48 – 1.22), suggesting there was a positive association between eastern spotted skunk detection and temperature.

I fit occupancy models for eastern spotted skunks using the top ranked detection model, temperature. The goodness of fit score for the global occupancy model was 1.45, so I used QAIC to rank my candidate models and inflated my standard error for the 85% confidence interval. There was some model selection uncertainty, with 2 models (distance to water and distance to water + percent ground cover by shrubs) in the 80% confidence set. The top-ranked occupancy model was distance to water (45% model weight; 1.19× more support than the next model) (Supplemental Material Table A1.5). In addition, the scaled effect size of the percent ground cover by shrubs in the distance to water + percent ground cover by shrubs model was not significant (-1.07; 85% confidence interval = -2.19 – 0.06). The scaled effect size of distance to water in the distance to water only model was -2.72 (85% confidence interval = -4.60 – -0.85), suggesting a negative association between eastern spotted skunk occupancy and increasing distance to water.

Masked Shrew

I observed 56 independent detections of masked shrews (*Sorex cinerus*) at 8 independent sites (Figure 1.3). The mean latency to initial detection was 87 days (range 2 – 203). The goodness of fit score for the global detection model was 0.43, indicating the model was overfit, but I still used AIC to rank my candidate detection models. There was model selection uncertainty, with 5 models (number of days the camera was knocked over, the null, temperature + number of days the camera was knocked over, slope, and *Peromyscus* relative activity index) in the 80% confidence set. The top-ranked detection model was identified the number of days the

camera was knocked over (33% model weight; 1.31× more support than the next model) (Supplemental Material Table A1.6). The scaled effect size of the number of days the camera was knocked over was -0.78 (85% confidence interval = -1.56 – -0.01), suggesting there was a negative association between masked shrew detection and the camera being knocked over.

I fit occupancy models for masked shrews using the top ranked detection model, the number of days the camera was knocked over. The goodness of fit score for the global occupancy model was 1.66, so I used QAIC to rank my candidate models and inflated my standard error for the 85% confidence interval. There was high model selection uncertainty, with 6 models (the null, *Peromyscus* relative activity index, the volume of coarse woody debris, distance to water, percent ground cover by shrubs, and litter depth) in the 80% confidence set. However, none of the effect sizes for the covariates in the models were significant, with all the 85% confidence intervals crossing zero. The top-ranked occupancy model was the null (34% model weight; 3.05× more support than the next model) (Supplemental Material Table A1.7). This suggests that masked shrew occupancy was constant across the study sites, that we did not have enough detections, or that we did not include an explanatory covariate for masked shrew occupancy in the model candidate set.

Pygmy Shrew

I observed 15 independent detections of pygmy shrews (*Sorex hoyi*) at 7 independent sites (Figure 1.4). The mean latency to initial detection was 99 days (range 47 – 168). The goodness of fit score for the global detection model was 0.85, indicating the model was overfit, but I still used AIC to rank my candidate detection models. There was model selection uncertainty, with 5 models (the null, the number of days the camera was knocked over, slope, *Peromyscus* relative activity index, and temperature) in the 80% confidence set. The top-ranked

detection model was the null, meaning detection was constant (28% model weight; 1.17× more support than the next model) (Supplemental Material Table A1.8).

The goodness of fit score for the global occupancy model was 0.91, so I used AIC to rank my candidate models. There was high model selection uncertainty, with 5 models (*Peromyscus* relative activity index, distance to water, forest type, the null, and distance to water + percent ground cover by shrubs) in the 80% confidence set. The top-ranked occupancy model was identified *Peromyscus* relative activity (48% model weight; 3.43× more support than the next model) (Supplemental Material Table A1.9). The scaled effect size of *Peromyscus* relative activity index was 1.10 (85% confidence interval = 0.14 – 2.06), suggesting a positive association between pygmy shrew and increasing *Peromyscus* activity.

Carolina Red-backed Vole

I observed 2 Carolina red-backed voles (*Clethrionomys gapperi carolinensis*) at 2 independent sites (24H7 and 24H9) (Figure 1.5). Both detections were from the first year of the study, and both occurred in winter months (November and December, respectively), with a mean latency to initial detection of 166 days (range 148 – 184). Two detections were not enough samples to fit occupancy models, but descriptively, the sites with detections were 400 meters apart, and overlapped with historic detections of Carolina red-backed voles on the Foothills Trail outside the Ellicott Wilderness Area (Laerm et al., 1995; NatureServe, 2024a). Both sites were high elevation, at 912 m and 916 m, with a North-facing aspect. They had 90% canopy cover (24H7 had 50% conifers in the canopy, and 24H9 had 30%), with the top three tree families at each site being similar; Aceraceae (maples), Fagaceae (beeches), and Pinaceae (pines). They had 6 cm of litter depth and 90-100% ground cover by shrubs, predominantly in the Ericaceae (blueberry) family. 24H7 had lower and 24H9 had higher volumes of coarse woody debris

compared to the mean volume of all sites. Both sites had higher average relative activity indices of *Peromyscus* species compared to the mean of the study area.

Woodland Jumping Mouse

I observed 1 woodland jumping mouse (*Napaeozapus insignis*) at 1 site (Figure 1.6). The detection was from the first year of the study and occurred in October, with a latency to detection of 166 days. A single detection was not enough samples to fit occupancy models, but descriptively, the site overlapped with the IUCN range for the species (Cassola, 2016a). The site was at a moderately high elevation, at 639 m, with a South-facing aspect. It had 80-90% canopy cover and 20% conifers in the canopy. The top three tree families present were Aceraceae (maple), Ericaceae (heath), and Magnoliaceae (magnolia). The litter depth was 7 cm and it had 70-80% ground cover by shrubs, predominantly in the Vitaceae (grape) family. The volume of coarse woody debris and average relative activity index of *Peromyscus* species were close to the means of all sites for each metric.

DISCUSSION

I detected all six target species within the two-year study period, confirming the persistence of these state-listed species in South Carolina. This demonstrates that, similar to previous studies that have trialed AHDriFT (Amber, Lipps, et al., 2021; Martin et al., 2017; White et al., 2023), the method can be an effective monitoring method for small mammals. Every occupancy covariate included in a target species' model confidence set also appeared in at least one other target species' confidence set, although the direction of the relationships varied and only distance to water, forest type, and *Peromyscus* relative activity were significant. Eastern spotted skunk occupancy had a negative association to distance to water, eastern woodrat

occupancy had a negative association to coniferous forest types, and pygmy shrew occupancy was positively associated with coniferous forest types and *Peromyscus* relative activity (Figure 1.7). Much of the previous knowledge of the distributions and habitat associations for these species in the state, particularly the less common masked shrew, pygmy shrew, Carolina red-backed vole, and woodland jumping mouse, are either descriptive accounts from very limited sample size (Laerm et al., 1995) or taken from studies in other regions of the species' range (Brannon, 2000, 2005; Brown, 1967; Ford & Rodrigue, 2001; Pauli et al., 2006). The masked shrew had last been sighted in two sites in the state during one study that took place more than thirty years ago (Laerm et al., 1995). The current study increased official records of masked shrew by 224% and pygmy shrew by 63%. For the Carolina red-backed vole and the woodland jumping mouse, these results contribute the first official record of these species in the state since 1995 and 1980 respectively (Laerm et al., 1995; NatureServe, 2024b). This study increased official detections in South Carolina of Carolina red-backed vole by 67% and woodland jumping mouse by 50%. There are more publications, and as such, more historic detections on eastern woodrats in South Carolina, but much of this focuses on coastal rather than montane populations and the majority focus on describing ectoparasites they carry rather than habitat preferences (Durden et al., 1997; Knowles & Burger, 2008; Oliver et al., 2000). This study increased official eastern woodrat detections by approximately 600%. The eastern spotted skunk in South Carolina has been studied more recently in high elevation forests, but these records for this species remain limited (Eng & Jachowski, 2019b, 2019a; Wilson et al., 2016). This study increased official detections by 121%.

The AHDriFT method produced sufficient data to fit occupancy models for four of my six target species: eastern woodrat, eastern spotted skunk, masked shrew, and pygmy shrew. My

findings support past investigations that suggest eastern spotted skunk occupancy is positively associated with proximity to drainages and other mesic sites (Eng & Jachowski, 2019b). These areas provide ample moisture and cover for thermoregulation, predator avoidance, and forage (Lesmeister et al., 2008). Previous studies have shown the eastern woodrat is a habitat generalist and opportunistic forager (Beckmann et al., 2001; McMurry et al., 1993). However, I found a negative association between eastern woodrat occupancy and coniferous dominated forest types. These results align with a Florida study for the *Neotoma floridana floridana* subspecies that reported they were associated with mesic hardwood sites (HaySmith, 1995). While I observed a positive association between pygmy shrew occupancy and *Peromyscus* relative activity index, the *Peromyscus* relative activity index was highly correlated with elevation. Thus, I hypothesize that the underlying relationship between pygmy shrew occupancy may be driven less by *Peromyscus* activity and more by high-elevation environments, which provide xeric, coniferous habitats that pygmy shrews can exploit (Baumgardner et al., 1999). Other studies from more northern populations suggest pygmy shrew is a generalist and can also occupy lower-elevation hardwood forests (DeGraaf et al., 1991).

For rarely detected species where I could not fit models to identify significant habitat associations (i.e., masked shrew, Carolina red-back vole, woodland jumping mouse), this study still contributed an update on their status and distributions. The detections I collected occurred near the historic detections for Carolina red-back vole and woodland jumping mouse, suggesting those populations have persisted and that the habitat in these areas continues to support these rare species. Because AHDriFT has proven successful at detecting these rare species in other regions (Amber et al., 2021b; Martin et al., 2017; White et al., 2023), to improve our understanding of habitat associations. Therefore, I recommend sampling more intensely with AHDriFT at both

historic and current detection locations. For rare species, sampling more sites for shorter durations would be more effective than only having a few sites for long-term monitoring (MacKenzie and Royle, 2005).

A limitation to AHDriFT in my study system was the disturbance to the deployments by black bears. American black bears (*Ursus americanus*) frequently disturbed the fence and camera traps, tearing down or completely destroying the AHDriFT systems in between site visits leading to data loss. Other unpublished projects have noted camera buckets knocked over due to wind or cows, but have not reported or quantified this effect of bears (J. Hogue-Manuel, Louisiana Department of Wildlife and Fisheries, personal communication). Of 59 sites, 95% had damage from bears at least once and sites lost an average of 22 trap nights (range 0 – 84) per 6 week survey occasion when combining the two cameras for each AHDriFT system. While larger small mammals including the eastern woodrat and eastern spotted skunk could still be observed when the camera was knocked over, for the smaller taxa, such as shrews, detection decreased as they are too small to be captured in the camera view when it is knocked over. To mitigate data loss due to disturbance from bears, I would recommend using AHDriFT in areas with lower bear densities or placing systems out when bears are less active, such as during winter hibernation.

I found that across most small target mammal species in this study, the chance of detecting an individual increased as temperature increased. Game cameras use temperature differentials between the animal and the background surface to trigger the infrared sensor, so generally there is a negative association between ambient temperature and wildlife detection (Swann et al., 2004). However, this effect is less pronounced in animals with low biomass and therefore a lower heat signature (Randler & Kalb, 2018). Furthermore, small mammals have high surface area-to-volume ratios which leads to increased heat loss, so to thermoregulate small

mammals decrease activity at lower ambient temperatures and increase activity as it gets warmer, which may explain the positive association I saw between small mammal detection and temperature (Conley & Porter, 1986). This suggests that while AHDriFT may be used to survey small mammals year-round, detection is likely to be higher when the animals are most active, such as in summer. This may also allow bears more opportunities to disturb the system, so depending on the population density of bears in the area managers may have to balance their survey efforts and how much time on AHDriFT system repair they want to invest with how many small mammal detections they want to collect. While I was generally able to identify my focal species in camera trap images from AHDriFT, *Sorex* shrews were difficult to identify to species due to close resemblance and size overlap. Additionally, shrews were prone to skirting the rim of the bucket, staying partially out of frame, which exacerbated this issue. A solution to mitigate this is to use a taller bucket (26.5 liter rather than 19 liter) so that shrews were in full view of the camera. However, even with this strategy, I would recommend caution when interpreting results from *Sorex* shrew observations, as some studies note more difficulty identifying down to the species level with the 26.5 liter bucket (Hoffman, n.d.). Future studies could explore if the 26.5 liter bucket and more sampling yields more results for habitat associations of these targets and other small mammals, particularly shrews.

Collectively, this study fills a critical knowledge gap about the current status of state-listed small mammals in South Carolina and proves that we can more effectively monitor them through innovative techniques and technology like AHDriFT. This technique reduces time and labor costs, and time demands from photo processing are likely to decline as deep learning and artificial intelligence continue to improve for programs like Wildlife Insights (Norouzzadeh et al., 2018). As such I recommend this method for other studies investigating the distribution and

habitat associations of small, cryptic wildlife. While this study provides baseline information on the distribution, relative occurrence, and habitat associations of these understudied small mammals, continued use of AHDriFT for long-term monitoring is needed to identify population trends and inform conservation and restoration goals (White et al. 2023). Ultimately, broader application of this technique will improve our ability to detect changes in cryptic species and support informed, proactive conservation and management decisions.

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TABLES

Table 1.1: Forest types found in the study area

FSVeg Forest Type	Total Area (square kilometers)	Percent of Polygon
White pine	22.82	14.63
White oak-northern red oak-hickory	20.89	13.40
Shortleaf pine-oak	19.60	12.57
White pine-cove hardwood	10.89	6.99
Chestnut oak-scarlet oak-yellow pine	10.00	6.41
White pine-upland hardwood	8.00	5.13
Upland hardwoods-white pine	6.54	4.20
Cove hardwood - white pine - hemlock	5.37	3.44
White oak-black oak-yellow pine	5.35	3.43
Shortleaf pine	4.99	3.20
Chestnut oak-scarlet oak	4.91	3.15
Loblolly pine	4.61	2.95
Pitch pine	4.19	2.69
Pitch pine-oak	3.52	2.26
Northern red oak-hickory-yellow pine	3.35	2.15
Yellow poplar-white oak-northern red oak	3.11	1.99
Southern red oak-yellow pine	2.65	1.70
Virginia pine	2.44	1.57
Hemlock-hardwood	2.36	1.52
Virginia pine-oak	1.82	1.16
Not Identified	1.67	1.07
White oak	1.64	1.05
Chestnut oak-White oak-Scarlet oak	1.44	0.92
White pine-hemlock	1.40	0.90
Loblolly pine-hardwood	0.61	0.39
Upland oak	0.50	0.32
Scarlet oak	0.42	0.27
Chestnut oak	0.36	0.23
Bottomland hardwood-yellow pine	0.31	0.20
Table Mountain-pine-hardwood	0.10	0.06
Yellow poplar	0.08	0.05

Table 1.2: Detection probability a-priori models. Covariate abbreviations: 'pero' *Peromyscus* relative activity index, 'temp' ambient temperature, 'dnf' number of days the camera was knocked over, 'season' categorical variable Spring/Summer/Autumn/Winter, 'slope' grade of the slope at the site.

Detection Probability Model	Eastern Woodrat	Eastern Spotted Skunk	Carolina Red-backed Vole	Pygmy Shrew	Masked Shrew	Woodland Jumping Mouse
p = β0 + β1(pero)	β1 < 0	β1 < 0	β1 < 0	β1 < 0	β1 < 0	β1 < 0
p = β0 + β1(temp)	β1 < 0	β1 < 0	β1 < 0	β1 < 0	β1 < 0	β1 < 0
p = β0 + β1(dnf)	β1 < 0	β1 < 0	β1 < 0	β1 < 0	β1 < 0	β1 < 0
p = β0 + β1(season)	Summer will increase detection	Summer will increase detection	Summer will increase detection	Summer will increase detection	Summer will increase detection	Summer will increase detection
p = β0 + β1(slope)	β1 < 0	β1 < 0	β1 < 0	β1 < 0	β1 < 0	β1 < 0
p = β0 + β1(temp) + β2(dnf)	β1 < 0, β2 < 0	β1 < 0, β2 < 0	β1 < 0, β2 < 0	β1 < 0, β2 < 0	β1 < 0, β2 < 0	β1 < 0, β2 < 0
p = β0 + β1(temp) + β2(dnf) + β3(pero) + β4(season) + β5(slope)	β1<0, β2<0, β3<0, β4<0, β5<0	β1<0, β2<0, β3<0, β4<0, β5<0	β1<0, β2<0, β3<0, β4<0, β5<0	β1<0, β2<0, β3<0, β4<0, β5<0	β1 < 0, β2 < 0, β3 < 0, β4 < 0, β5 < 0	β1<0, β2<0, β3<0, β4<0, β5<0
p = β0						

Table 1.3: Occupancy probability a-priori models and the hypothesis they test. Covariate abbreviations: 'cft' canopy forest type (percent of conifers), 'ld' litter depth, 'dw' distance from water, 'ps' percent ground cover by shrubs, 'cwd' the volume of coarse woody debris, 'pero' *Peromyscus* relative activity index.

Hypothesis	Model	Eastern Woodrat	Eastern Spotted Skunk	Carolina Red-backed Vole	Pygmy Shrew	Masked Shrew	Woodland Jumping Mouse
Occupancy is associated	ψ = β0 + β1(cft) + p	β1=0	β1=0	β1>0	β1=0	β1>0	β1>0
with food and water	ψ = β0 + β1(ld) + p	β1=0	β1=0	β1<0	β1>0	β1>0	β1=0
availability	ψ = β0 + β1(dw) + p	β1=0	β1<0	β1<0	β1=0	β1<0	β1<0
	ψ = β0 + β1(ps) + p	β1>0	β1>0	β1>0	β1>0	β1>0	β1>0
Occupancy is associated	ψ = β0 + β1(cwd) + p	β1>0	β1>0	β1>0	β1>0	β1>0	β1>0
with shrubby vegetation	ψ = β0 + β1(ps) + β2						
and coarse woody debris	(cwd) + p	β1>0, β2>0	β1>0, β2>0	β1>0, β2>0	β1>0, β2>0	β1>0, β2>0	β1>0, β2>0
Occupancy is associated							
with Peromyscus activity	ψ = β0 + β1(pero) + p	β1=0	β1=0	β1<0	β1<0	β1=0	β1<0
Occupancy is associated							
with ground cover and	ψ = β0 + β1(ld) +						
forage availability	β2(ps) + p	β1=0, β2>0	β1=0, β2>0	β1<0, β2>0	β1>0, β2>0	β1>0, β2>0	β1=0, β2>0
Occupancy is associated							
with ground cover and	ψ = β0 + β1(dw) +						
water availability	β2(ps) + p	β1=0, β2>0	β1<0, β2>0	β1<0, β2>0	β1=0, β2>0	β1<0, β2>0	β1<0, β2>0
	ψ = β0 + β1(dw) +						
	β2(ps) + β3(ld) +				β1=0, β2>0,	β1<0, β2>0,	
	β4(cwd) + β5(cft) +	β1=0, β2>0,			β3>0, β4>0,	β3>0, β4>0,	
Global	β6(pero) + p	β3=0, β4>0, β5=0, β6=0	β1<0, β2>0, β3=0, β4>0, β5=0, β6=0	β1<0, β2>0, β3<0, β4>0, β5>0, β6<0	β5=0, β6<0	β5>0, β6=0	β1<0, β2>0, β3=0, β4>0, β5>0, β6<0
Null	ψ = β0 + p						

Table 1.4: Species detection summary

Species	AHDriFT	External Camera	Total per Species	Target Species?
Peromyscus spp.	3506	568	4074	
Eastern Gray Squirrel (Sciurus carolinensis)	881	1371	2252	
American Black Bear (Ursus americanus)	566	469	1035	
Eastern Chipmunk (Tamias striatus)	357	306	663	
Nine-banded Armadillo (Dasypus novemcinctus)	59	271	330	
Northern Raccoon (Procyon lotor)	79	225	304	
Eastern Woodrat (Neotoma floridana haematoreia)	108	192	300	Yes
Virginia Opossum (Didelphis virginiana)	131	153	284	
Mouse	188	37	225	
Sylvilagus spp.	15	119	134	
Eastern Spotted Skunk (Spilogale putorius)	30	52	82	Yes
Southern Flying Squirrel (Glaucomys volans)	23	59	82	
Five-lined Skink (Eumeces fasciatus)	79	0	79	
Coyote (Canis latrans)	29	37	66	
Northern Short-tailed Shrew (Blarina brevicauda)	62	0	62	
Bobcat (Lynx rufus)	15	44	59	
Smoky Shrew (Sorex fumeus)	56	3	59	
Masked Shrew (Sorex cinereus)	56	0	56	Yes
White-tailed Deer (Odocoileus virginianus)	22	32	54	
Feral pig/hog/swine (Sus scrofa)	7	46	53	
Shrew	33	1	34	
Woodland Vole (Microtus pinetorum)	20	0	20	
Southern Gray-cheeked Salamander (Plethodon metcalfi)	2	0	2	
American Pygmy Shrew (Sorex hoyi)	15	0	15	Yes
Common Gray Fox (Urocyon cinereoargenteus)	0	12	12	
Long-tailed Weasel (Mustela frenata)	3	7	10	
Striped Skunk (Mephitis mephitis)	1	9	10	
Common Garter Snake (Thamnophis sirtalis)	9	0	9	
Rat Snake (Elaphe obsoleta)	8	1	9	
Blue Ridge Two-lined Salamander (Eurycea wilderae)	1	0	1	
Copperhead (Agkistrodon contortrix)	7	0	7	
Northern Slimy Salamander (Plethodon glutinosus)	5	0	5	
Eastern American Toad (Bufo americanus)	5	0	5	
Racer (Coluber constrictor)	3	1	4	
Eastern Box Turtle (Terrapene carolina)	2	0	2	
Hispid Cotton Rat (Sigmodon hispidus)	2	0	2	
Least Shrew (Cryptotis parva)	2	0	2	
Red Squirrel (Tamiasciurus hudsonicus)	0	2	2	
Salamander	2	0	2	
Southern Appalachian Salamander (Plethodon oconaluftee)	1	0	1	
Carolina Red-backed Vole (Clethrionomys gapperi)	2	0	2	Yes
Black-chinned Red Salamander (Pseudotriton ruber)	1	0	1	
Brown Snake (Storeria dekayi)	1	0	1	
Corn Snake (Elaphe guttata)	1	0	1	
Eastern Harvest Mouse (Reithrodontomys humilis)	1	0	1	
Green Anole (Anolis carolinensis)	1	0	1	
Red Fox (Vulpes vulpes)	0	1	1	
Scarlet Kingsnake (Lampropeltis elapsoides)	1	0	1	
Woodchuck (Marmota monax)	0	1	1	
Woodland Jumping Mouse (Napaeozapus insignis)	1	0	1	Yes

Table 1.5: Target species detection summary by season

Species	**Summer**	**Autumn**	**Winter**
American Pygmy Shrew (*Sorex hoyi*)	2	8	5
Eastern Spotted Skunk (*Spilogale putorius*)	45	36	1
Eastern Woodrat (*Neotoma floridana haematoreia*)	183	107	10
Masked Shrew (*Sorex cinereus*)	4	46	6
Carolina Red-backed Vole (*Clethrionomys gapperi carolinesis*)	0	1	1
Woodland Jumping Mouse (*Napaeozapus insignis*)	0	1	0

FIGURES

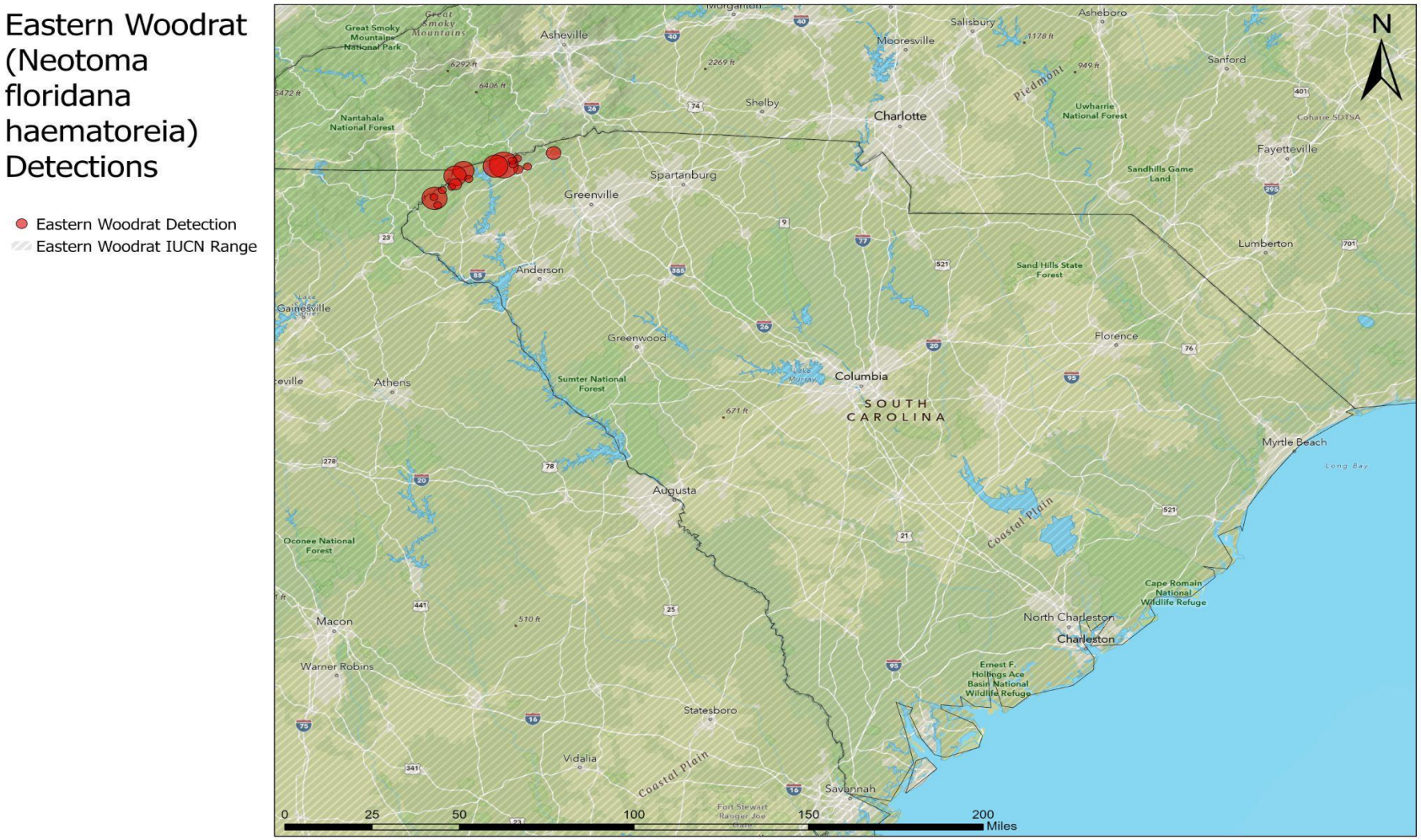

Figure 1.1: Map of all independent observations of eastern woodrats (*Neotoma floridana haematoreia*) at study sites from May 2024 – February 2026 with the IUCN range. Larger circles indicate multiple observations at the same site.

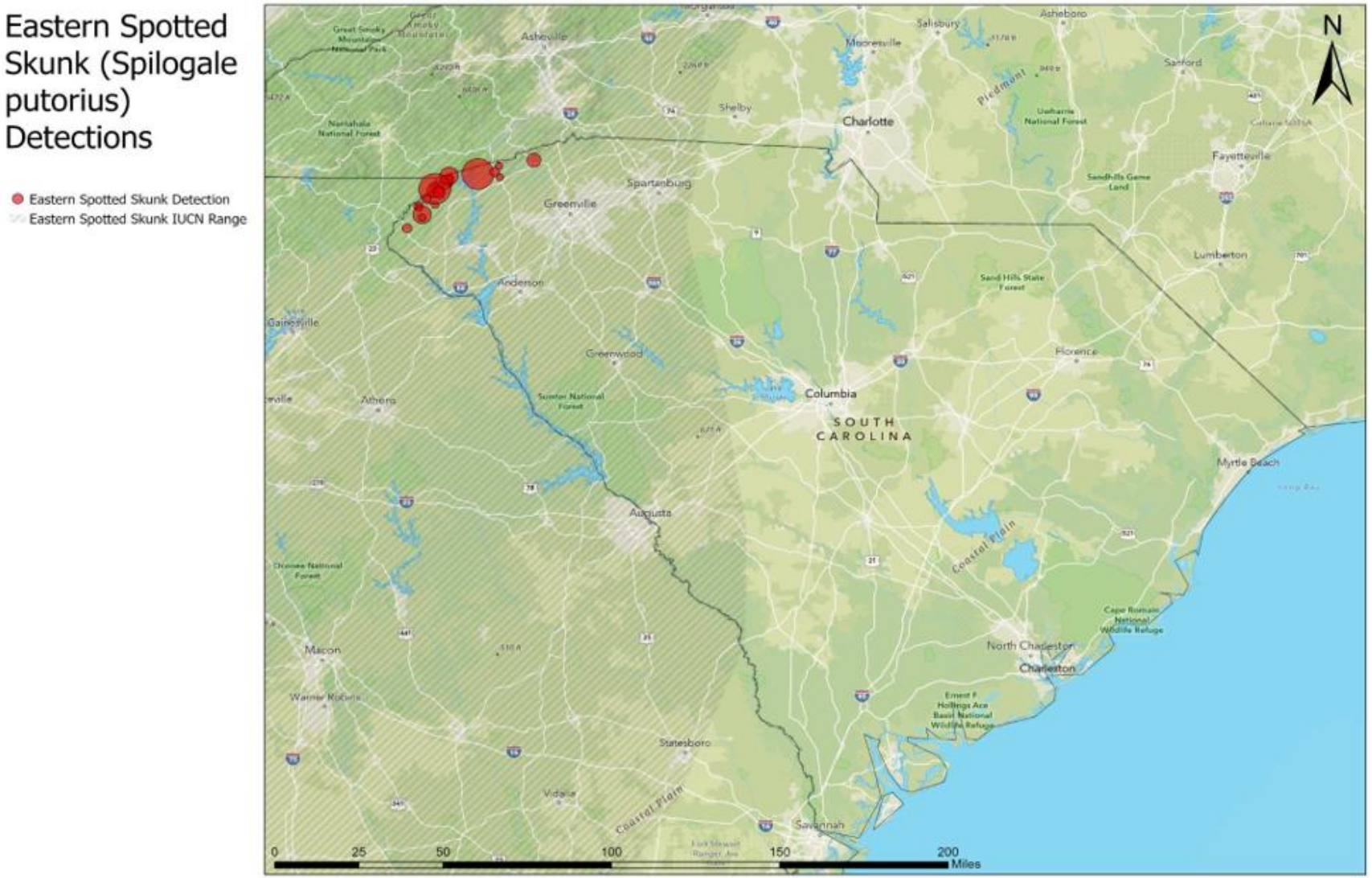

Figure 1.2: Map of all independent observations of eastern spotted skunk (*Spilogale putorius*) at study sites from May 2024 – February 2026 with the IUCN range. Larger circles indicate multiple observations at the same site.

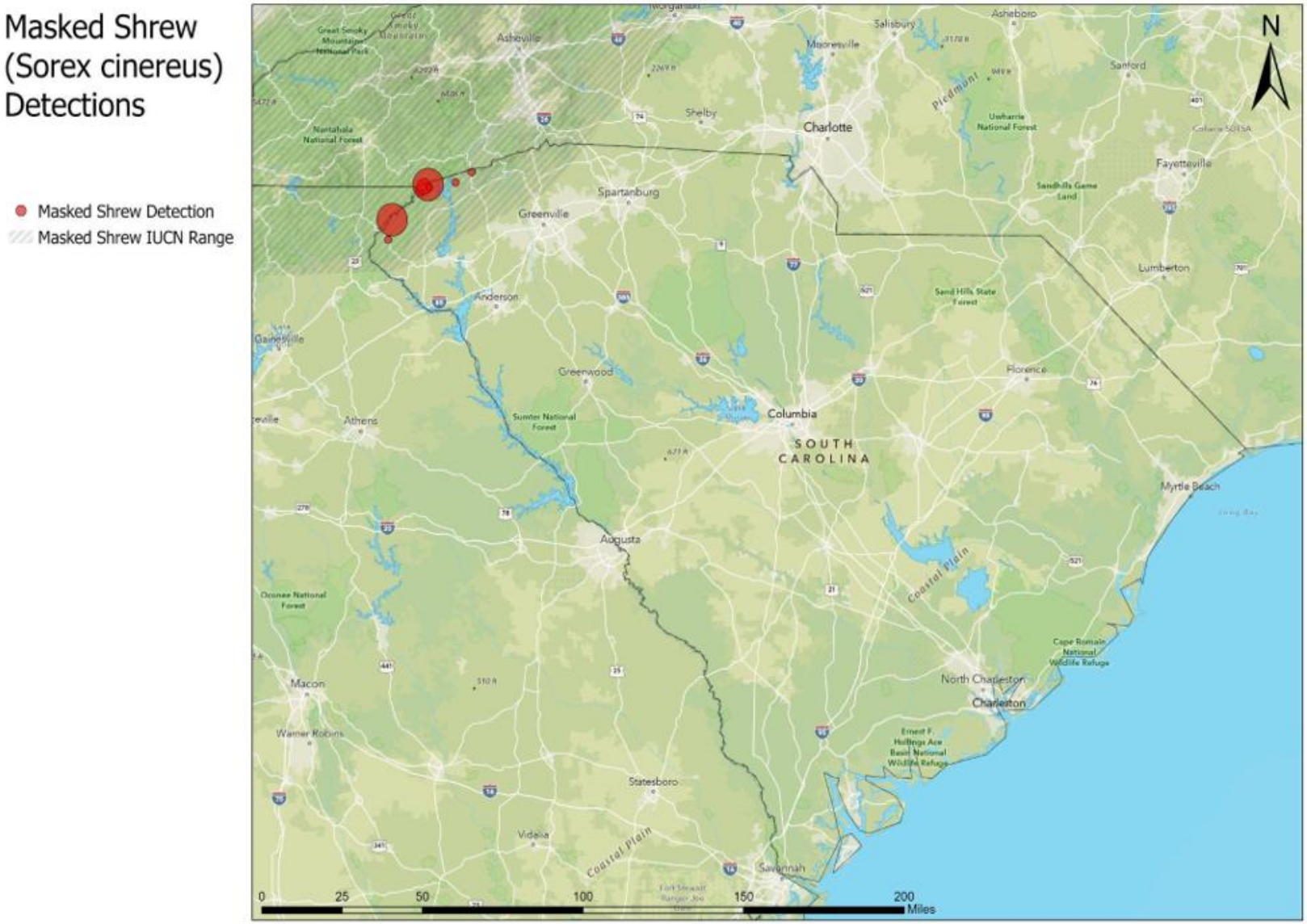

Figure 1.3: Map of all independent observations of masked shrew (*Sorex cinereus*) at study sites from May 2024 – February 2026 with the IUCN range. Larger circles indicate multiple observations at the same site.

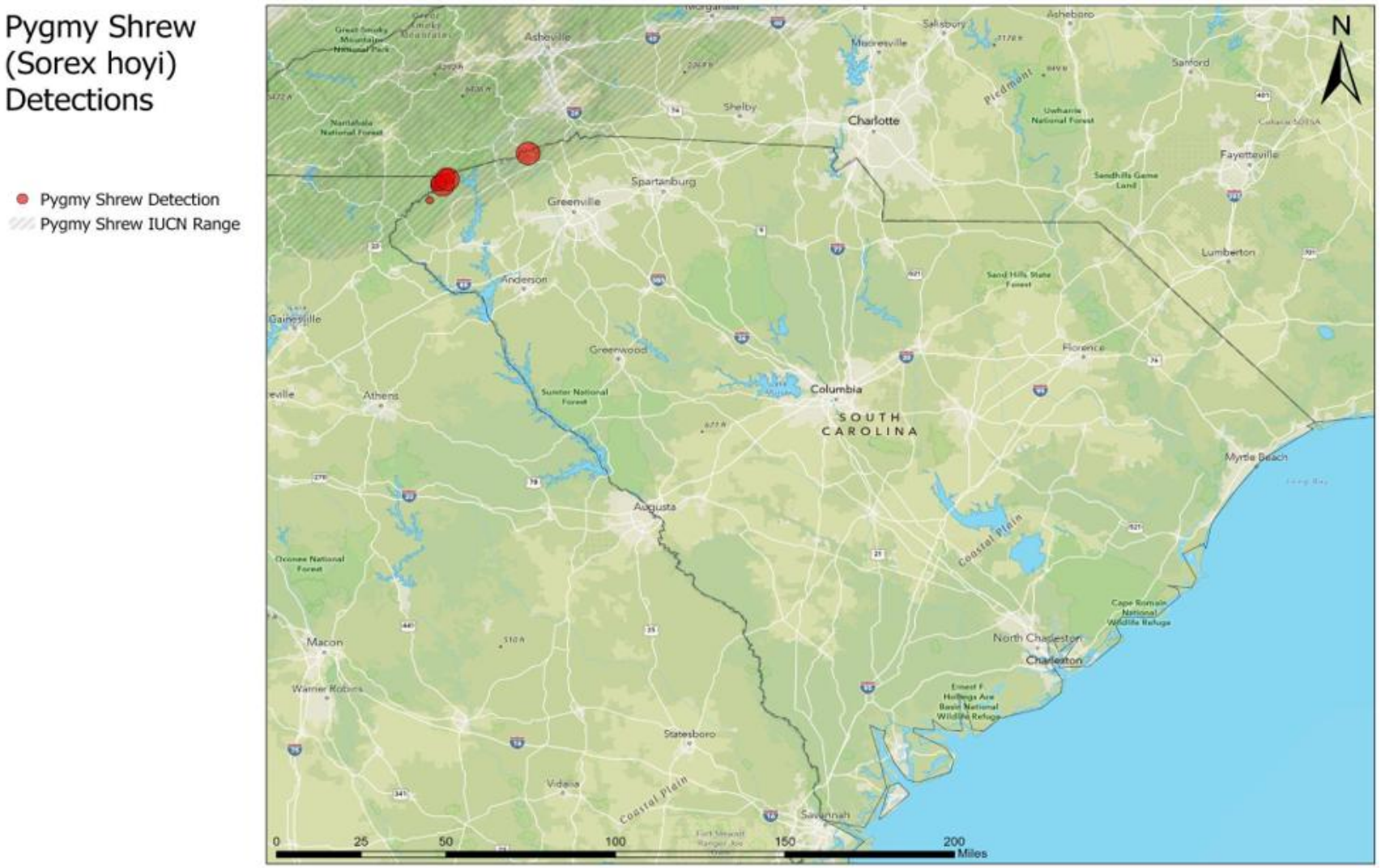

Figure 1.4: Map of all independent observations of pygmy shrew (*Sorex hoyi*) at study sites from May 2024 – February 2026 with the IUCN range. Larger circles indicate multiple observations at the same site.

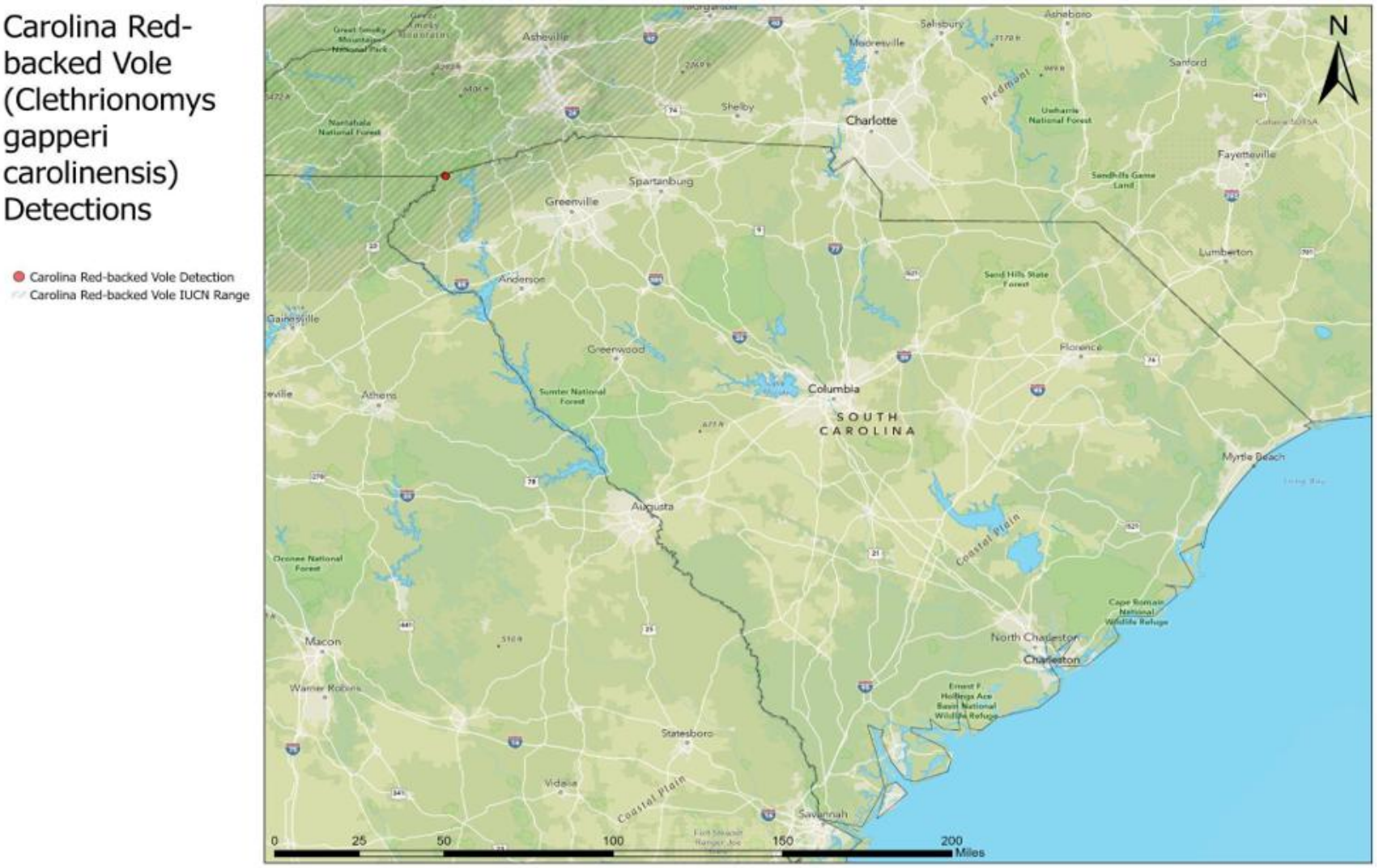

Figure 1.5: Map of all independent observations of Carolina red-back vole (*Clethrionomys gapperi carolinensis*) at study sites from May 2024 – February 2026 with the IUCN range.

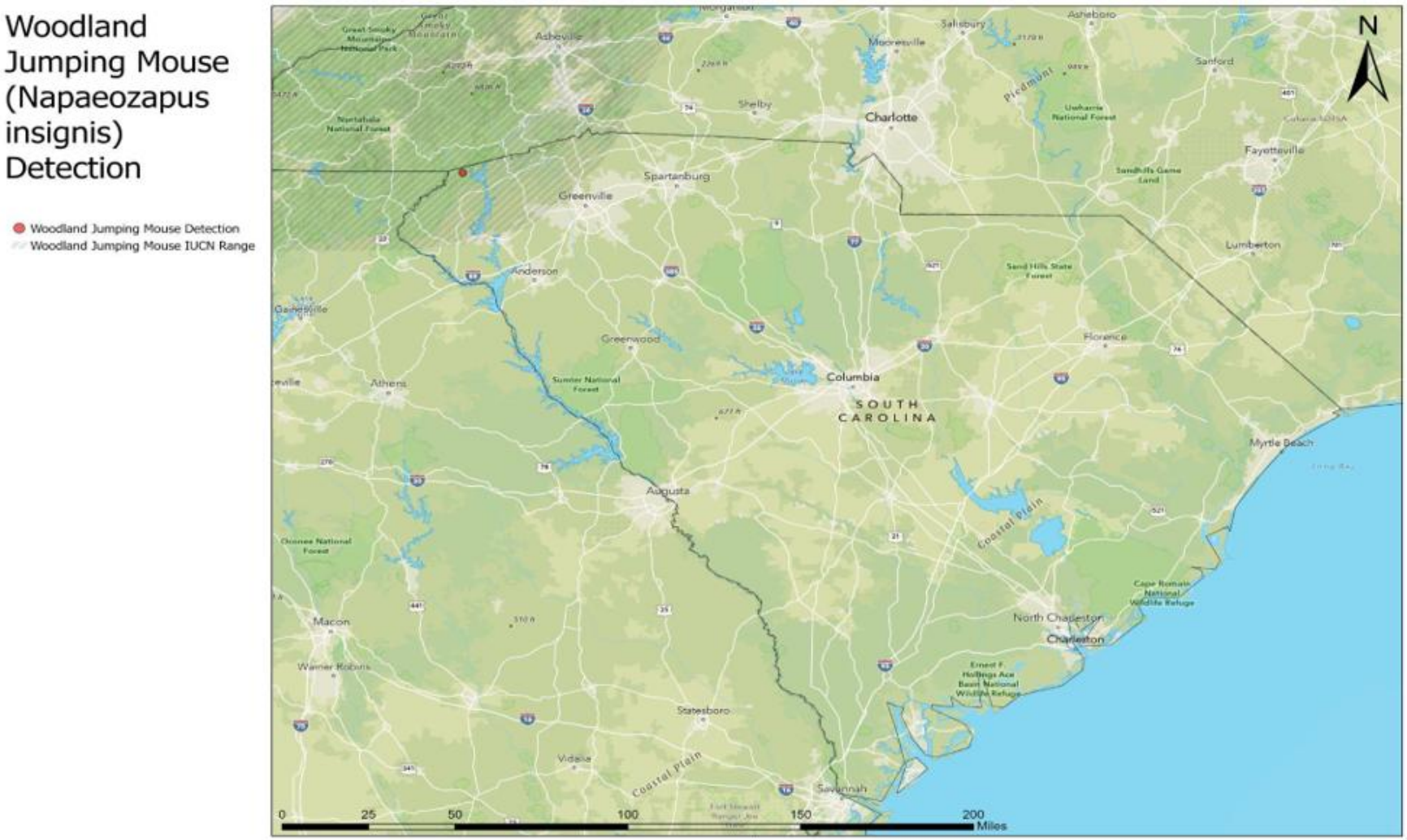

Figure 1.6: Map of the independent observation of a woodland jumping mouse (*Napaeozapus insignis*) at study sites from May 2024 – February 2026 with the IUCN range.

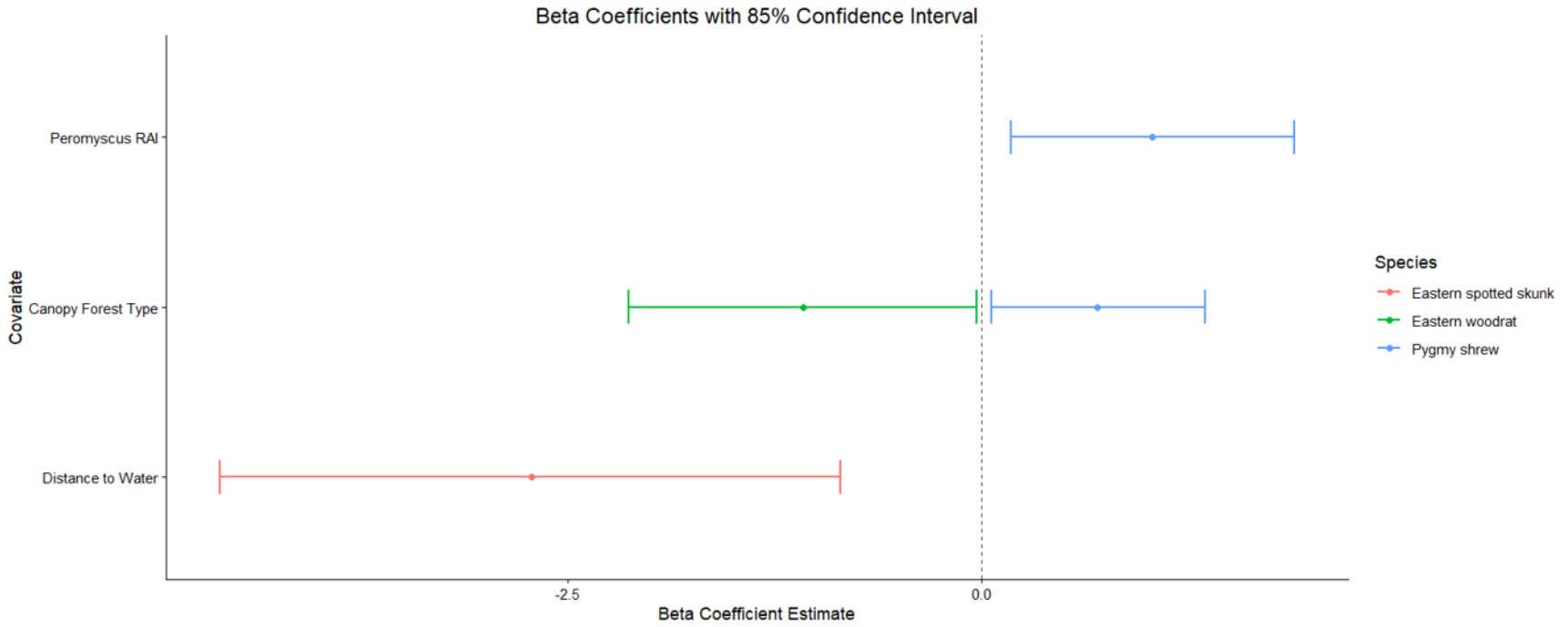

Figure 1.7: β coefficients with 85% confidence intervals for significant occupancy model covariates. “Canopy Forest Type” is percent of conifers in the canopy, so a negative coefficient would indicate hardwood-dominated sites and a positive coefficient would indicate more coniferous forest types.

CHAPTER TWO: SMALL MAMMALS AS INDICATORS OF BIODIVERSITY IN THE BLUE RIDGE MOUNTAINS

INTRODUCTION

Threats to biodiversity such as climate change and habitat loss pose formidable challenges to conservationists today, with 28% of all assessed species and 26% of mammals on the IUCN Red List of Threatened Species (*The IUCN Red List of Threatened Species*. (n.d.)). Conservation efforts do not have the time or resources to conserve these 45,000+ species of flora and fauna, so conservation practitioners often prioritize sentinel species that indicate ecosystem condition, support key ecological functions, protect co-occurring species, or are charismatic enough to attract public support (Clark-Wolf et al., 2024; Zacharias and Roff, 2001). Examples include the use of sabine shiner (*Notropis sabinae*) as a sentinel species for stream diversity and ecosystem health in Texas (Antoniazzi et al., 2023), or the use of butterfly (Lepidoptera) communities as sentinels for biodiversity of cryptic species in the Iberian Peninsula (Dincă et al., 2015).

The most effective sentinels have direct trophic links through bottom-up regulation, high diversity, and short generation times (Clark-Wolf et al., 2024; Marneweck et al., 2022), characteristics that are well represented in small mammal communities (Ayomiposi Ayodele, 2025; Linzey, 2016). Small mammals influence plant communities through seed and seedling foraging that suppresses recruitment and establishment (Byrom et al., 2014; Bricker et al., 2010; Hogan et al., 2024; Maron et al., 2010; Meserve et al., 2003; Schnurr et al., 2004), while also serving as an important food resource that transfers energy through food webs and can regulate predator populations through bottom-up processes (Hogan et al., 2024; Krebs et al., 2019; Byrom et al., 2014; Korpimäki et al., 1991; Letnic & Dickman, 2010; Norrdahl & Korpimäki, 2002;

Villar et al., 2014). Conversely, predation can regulate small mammal populations, and some systems alternate between bottom-up and top-down control (Levi et al., 2012; Maron et al., 2010; Ruxton & Lima, 1997; Therrien et al., 2014; Butler et al., 2023; Byrom et al., 2014; Elmhagen and Rushton, 2007; Etcheverry et al., 2004; Letnic et al., 2011; Letnic & Dickman, 2010; M. Lima et al., 2002; Meserve et al., 2003). Their central role in food webs, functional diversity as granivores and insectivores, and importance as prey suggest that small mammals may serve as effective sentinels for multiple trophic levels and ecosystem conditions (Marneweck et al., 2022; Linzey 2016; Ayomiposi Ayodele, 2025). The potential value of small mammals as sentinels is especially relevant in the southern Blue Ridge Mountains, a biodiversity hotspot (Jenkins et al., 2015) and conservation priority due to its biodiversity and endemism (Myers, 1988). The region supports diverse communities of invertebrates, herpetofauna, and small predators, many of which are of conservation concern, including 12 state-listed insects, 10 state-listed herpetofauna, and 15 state-listed small mammals (excluding bats) and small mammalian predators (South Carolina Department of Natural Resources, 2020; South Carolina Department of Natural Resources, 2025). If small mammals reliably indicate the status of these other trophic levels, land managers could streamline biodiversity monitoring by using them as indicators of broader animal communities.

In this study, I evaluated the extent to which small mammals serve as sentinels for the diversity and relative activity of lower trophic levels (invertebrates), equivalent trophic levels (herpetofauna), and higher trophic levels (small predators) in the southern Blue Ridge Mountains. Small predators included species that prey on small mammals, such as coyotes, foxes, bobcats, snakes, raccoons, opossums, skunks, and weasels. My first objective was to determine whether small mammal diversity is a good indicator of the diversity of invertebrates,

herpetofauna, and small predators. I predicted that greater small mammal diversity would be associated with greater predator diversity because niche differentiation among prey provides opportunities for predators with different specializations and predation strategies (Dammhahn et al., 2013; Fox, 2004; Therrien et al., 2014). I also predicted positive associations between small mammal diversity and the diversity of invertebrates and herpetofauna because these groups may share habitat preferences (Aslan et al., 2015).

In this study my goal was to evaluate the extent to which small mammals are good sentinels for the diversity and abundance of lower trophic levels (invertebrates), equivalent trophic levels (herpetofauna) and higher trophic levels (small predators) in the southern Blue Ridge Mountains. Small predators in this context are animals that prey on small mammals, including coyotes, foxes, bobcats, snakes, raccoons, opossums, skunks, and weasels. My first objective was to evaluate support for the hypothesis that small mammal diversity is a good indicator of the diversity of invertebrates, herpetofauna, and small predators. I predicted that more diverse small mammal populations are associated with more diverse predator populations, as niche differentiation allows a larger variety of small mammals to open more opportunities for predators with different specializations or predation methods to take advantage (Dammhahn et al., 2013; Fox, 2004; Therrien et al., 2014). I also predicted small mammal diversity is associated with diversity of invertebrates and herpetofauna as they may have shared habitat preferences (Aslan et al., 2015). My second objective was to investigate relationships between the relative activity of small mammals and other trophic levels while accounting for habitat conditions. I hypothesized that small predators track small mammal use of sites (Knop et al., 2014; Korpimäki et al., 1991; Norrdahl & Korpimäki, 2002) and predicted that predator relative activity, particularly that of mustelids and snakes, would increase with small mammal activity (Pivorun et

al., 2006; Sundall et al., 2013; Greenhorn et al., 2021), especially for mustelids during winter when alternative prey are limited (Korpimäki et al., 1991; Norrdahl & Korpimäki, 2002). I further predicted that sites with high small mammal activity would support more active soil faunal communities, resulting in greater herpetofauna activity because both groups depend on water availability and soil moisture. Specifically, I expected shrew activity to be positively associated with salamander activity (Laerm et al., 1995; P. Brannon, Highlands Biological Station, personal communication). Because herpetofauna and small mammals both respond positively to coarse woody debris in this region (Owens et al., 2008; Loeb, 1999), and shrews depend on insects as a food source (Meserve et al., 2003; Vandegehuchte et al., 2017), I also predicted positive relationships between small mammal activity and invertebrate abundance (Brown 1967).

My third objective was to evaluate how microhabitat conditions important to the southern Appalachian ecosystem influence overall small mammal and shrew relative activity. Leaf litter plays an important role in retaining moisture, regulating soil pH, facilitating nutrient cycling, and providing cover and forage for small mammals and herpetofauna (Brannon, 2000; Degraaf & Rudis, 1990; Kaminski et al., 2007; Sayer, 2006). I therefore predicted that small mammal activity would increase with moist environmental conditions, abundant leaf litter, and greater forage availability (Bovendorp et al., 2020; Byrom et al., 2014; Letnic & Dickman, 2010; Lima et al., 2002, 2003; Lima & Jaksic, 1998). Because species such as red-backed voles, woodland jumping mice, pygmy shrews, and eastern woodrats consume both seeds and insects (Meserve et al., 2003; Vandegehuchte et al., 2017), I also predicted positive associations between small mammal activity and seed abundance (Orrock et al., 2003; Brannon 2005; Pivorun et al., 2006; Campbell et al., 2010; Linzey, 2016). Focusing specifically on shrews because of their

insectivorous niche, I predicted that shrew activity would be positively associated with invertebrate abundance (Brown 1967) and litter depth (Brown, 1967; Goertz, 1970; Linzey, 2016; Pivorun et al., 2006), conditions that also benefit salamanders and other litter-associated fauna (Degraaf & Rudis, 1990; Paradise, 2004). Collectively, these results can inform land managers whether small mammals can serve as effective sentinels in this region.

METHODS

Study Area

This study occurred in the Blue Ridge Mountain Range in South Carolina, in the northwest corner of the state, bordering Georgia and North Carolina, in Oconee and Pickens counties. I focused on high elevation Appalachian oak forests and set the low elevation threshold at 426 m based on the lower elevation limit of state-listed, high elevation obligate small mammal species in South Carolina (South Carolina Department of Natural Resources, 2020). This allowed me to focus the study on the high-elevation forest communities that are vulnerable to climate change (Diaz et al., 2003). The mean elevation in the study area is 737 m (range 590 m – 1018 m).

Site Selection

I selected study sites for AHDriFT placement using three criteria. First, to ensure access, I selected only publicly owned federal and state lands, including state parks, US Forest Service lands, and DNR lands. Second, sites were required to have reasonable access by buffering roads and trails by 500 m, while excluding areas within 200 m of roads to minimize traffic disturbance. Third, I focused on areas with coarse woody debris and dense understory vegetation, habitat features that are most abundant in drainages and are preferred by many small mammal species,

by buffering streams by 50 m (Bowman et al., 2001; Brannon, 2005; Ford & Rodrigue, 2001; Gooley & Schauber, 2018; Loeb, 1999; Pauli et al., 2006). I used an elevation threshold of 762 m (2500 ft) as a proxy for high-elevation, xeric or coniferous habitats used by other small mammal species (South Carolina Department of Natural Resources, 2020).

I stratified the portion of the study area meeting my three criteria into low (426 – 610 m), medium (610 – 762 m), and high (>762 m) elevation classes. Low and medium elevations were within the drainage buffer, while high elevations occurred on ridgetops. This approach ensured sampling across the full spatial and elevational range, with at least one-third of sites located on ridgetops and steep slopes. For each of these elevation categories I randomly generated 15 points each year, buffered by 805 meters (0.5 miles) to reduce spatial autocorrelation. Around the Ellicott Wilderness Area, some points were spaced 400-480 meters apart to increase sampling intensity due to a high number of historical detections of rare small mammal species. I selected 10 sites of the 15 optional points for each elevation category for each of the two years of study. This meant my total sample size was 60 sites, 30 per year. One of the sites was destroyed by Hurricane Helene in September 2024, which brought my sample size down to 59.

AHDriFT System

I placed an AHDriFT system at the nearest accessible point within 300 meters to the centroid of each study site, selecting the exact deployment location based on the availability of understory cover and coarse woody debris, but also space to lay out the 10 m long drift fence. Following the design of Martin et al. (2017), I installed a 10 meter long by 1 meter tall silt fence supported by 5 evenly spaced stakes. I brushed away litter and duff along the fence line but replaced it after staking in the drift fence so that the fence was partially buried, using large sticks and logs to anchor the fence down. At each end I placed a 19 liter overturned bucket with a 6-cm

by 6-cm box shaped hole carved into opposite sides of the bucket, ensuring the holes were placed parallel to the drift fence so that the animals were guided by the drift fence into the hole. I placed 2.5-cm by 10-cm boards at an angle towards the inside of the bucket to further funnel the animal into the bucket. On the underside of the bucket, I bolted a Browning TrophyCam HD trail camera facing the ground, so that the animals entering the bucket had their photo taken from above.

I modified the cameras by attaching a 4.00 diopter reading glass lens to help the camera focus the correct distance (30 cm) to the ground (Meek and Cook 2022), using caulk around the lens to avoid condensation. I also placed 20 layers of masking tape over the infrared flash to reduce photograph washout. I set the camera to take 3 image-burst photos when triggered, with a 3 second delay between trigger events. To monitor larger animals that visit the fence but didn't go in the buckets, I placed a third camera on a nearby tree facing the system.

The system once placed was left out for one year. I performed a first check when the system had been out for one week to make sure it was operating correctly, and then checked the system once a month to switch out the SD cards and replace the batteries. After the first year, the sites were moved to 30 new locations, and monitored for another year with monthly checks.

Soil Community Quadrants

I sampled invertebrates and herpetofauna at each AHDriFT site (10 meter radii around the center of the fence). The 10 meter circle plot was divided into four equal quadrants. The drift fence marked the division into two equal halves and a line perpendicular to the drift fence, going through the midpoint of the fence, further divided the site into quarters (Figure 2.1). Each quarter, or quadrant, corresponded to a particular sample. To sample herpetofauna, such as salamanders or lizards, I placed coverboards in Quadrants 1 and 3 at each site. The coverboards

were 60-cm by 60-cm, 1-cm thick, untreated plywood. I placed them randomly within the quadrant, wherever the ground was clear and flat enough to put them. I checked coverboards at each visit and any herpetofauna found underneath them was identified to species and classified as adult or juvenile (Pittman and Dorcas 2006). This supplemented the herpetofauna detections gathered from camera traps at the site.

I sampled invertebrates by taking a 25-cm by 25-cm leaf litter sample (Sandler et al., 2010) from Quadrant 2 at each site, prioritizing sampling an area where the ground was clear of shrubbery to get an uninterrupted square of leaf litter. I sampled the invertebrates twice per site, once in summer and once in winter. I bagged the litter sample in pillowcases and processed it through a Berlese Tullgren system in a lab. The Berlese Tullgren system consisted of a funnel with an incandescent light bulb on top and a beaker of 95% ethanol on the bottom. Any invertebrates in the litter sample were driven away from the light and heat of the light bulb and fell down the bottom of the funnel, where they were caught in the beaker of alcohol and preserved for identification (Smith et al., 2008). The sample processed through the funnel for 48 hours, and I preserved the sample in the ethanol after that. I then identified the preserved invertebrates to order and counted them to estimate abundance (Supplemental Material Table B2.1).

To estimate seed availability, I did seed counts in Quadrant 4 at each site within a random 1 m^2 area. I sampled each site twice, once in summer and once in winter, to evaluate summer and winter food availability for granivores. I used a trowel to move the litter down to the topsoil onto a tarp (Williams et al., 2019), where I sorted through it by hand to find and count seeds to estimate abundance.

Analysis

To address my first objective that small mammal diversity can be used as an indicator of diversity of other taxa, I used generalized linear models (GLMs) with a gaussian distribution from base R (R Core Team 2023) (Aslan et al., 2015; Grass et al., 2015). I calculated species richness, Shannon-Wiener (*H'*), and Simpson's (*1 – D*) diversity indices for each of my taxonomic groupings as these are the most commonly used diversity indices in ecology (Hopton et al., 2017). I treated all photographs of *Peromyscus* and *Sylvilagus*, which could not be identified to species, as if each genus was one species. As there are only two species for each of these genera (*Peromyscus maniculatus* and *Peromyscus leucopus*; *Syvilagus obscurus* and *Sylvilagus floridanus*) in the area, this did not strongly impact the diversity metrics. Simpson's and Shannon's indices were highly correlated, so I only modeled Shannon's index and species richness. I evaluated diversity of each taxon (invertebrates, herpetofauna, and small predators) as the response variable, and small mammal diversity as a predictor variable, along with models that included effects of habitat covariates (distance to water and percent ground cover by shrubs) to control for habitat variance (Table 2.1). I ranked models using AICc and examined the significance of effect of covariates within the top-ranked model by plotting model residuals and based on if the 85% confidence intervals of β estimates overlapped zero (Arnold 2010).

To address my second hypothesis that small mammal relative activity positively predicted the relative activity of invertebrates, herpetofauna, and small predators, I used generalized linear models (GLMs) with a negative binomial distribution from the R package *MASS* (Venables & Ripley, 2002) to evaluate how small mammal relative activity ranked compared to other explanatory models. I used the same habitat covariates as the diversity models (Table 2.1). As the camera data do not identify individuals, I used detections of species that were

at least 60 minutes apart for each species to calculate relative activity at each site (Amber, Jr, et al., 2021; Amber, Lipps, et al., 2021; Martin et al., 2017; White et al., 2023). I ranked models using AICc and evaluated quality of the top-ranked model by plotting model residuals to test for dispersion, uniformity, and outliers using the R package *DHARMa* (Hartig, 2024). I examined the significance of covariates within the model based on if the 85% confidence intervals of β estimates overlap zero.

To address my third objective to investigate the extent to which small mammal relative activity is a good sentinel for important microhabitat conditions in the region, I used N-mixture models. This method allowed me to estimate detection probability (p) and mean abundance (λ) of a population (Madsen & Royle, 2023). In this analysis, I interpreted λ as relative activity rather than true abundance, and in calculating this value I summed all small mammal detections across 42-day survey occasions. I developed 4 detection models with single covariates: average maximum daily temperature (*temp*), number of days the camera was knocked over (*dnf*), slope grade at the site (*slope*), and season (*season*). I used climate data from the National Weather Service to calculate the average maximum daily temperature for each site and survey occasion by using the *extract* tool from the R package *terra* (Hijmans, 2025) and *st_nearest_feature* from the R package *sf* (Pebesma, 2018) to find the closest weather tower for each site by geometric distance. The available towers were Caesars Head, Highlands, Jocassee 8 WNW, Lake Toxaway 2 SW, Long Creek, Pickens, Table Rock Reservoir, and Walhalla. To calculate the total number of days the camera was knocked over at each site for each survey occasion, I added the number of days the bucket was down for both cameras at each site. This gave a total number between 0 and 84 for the 42 day survey occasion. I used the *extract* tool from the R package *terra* and a DEM layer (Hijmans, 2025; *WorldElevation/Terrain (ImageServer)*, n.d.) to calculate the slope

grade at each site. I scaled all continuous covariates prior to model fitting. I used season as a categorical covariate with the categories Spring, Summer, Autumn, and Winter. I assigned the season for each survey occasion from the month of the midpoint date of the occasion, with March - May as Spring, June - August as Summer, September - November as Autumn, and December - February as Winter. However, we did not have data for more than 6 occasions, so there were no Spring samples. I also developed sub-global models: *temp* + *dnf*, to look at an additive combination of temperature and days knocked over, and *temp* * *dnf*, to look at an interactive combination of temperature and days knocked over. In addition to fitting models for all small mammals, I partitioned my data and fit shrew only N-mixture models. For this candidate model set I included a detection model for *Peromyscus* relative activity index (*pero*). For both groupings, I checked the goodness of fit (c-hat score) for the global model and ranked models using Akaike's Information Criterion adjusted for small sample size (AICc), and carried over the top-ranked detection probability model for each species to fit N-mixture models in a two-step process.

I created 7 a-priori N-mixture models, and 4 of those models use single habitat covariates (Table 2.2). I predicted small mammal relative activity would be affected by moisture availability (*dist_water*), seed availability (*seed_count*), invertebrate availability (*invert_count*), and litter depth (*ld*) (Brannon, 2005; Campbell et al., 2010; Linzey, 2016; Meserve et al., 2003; Orrock et al., 2003; Pivuron et al. 2006; Vandegehuchte et al., 2017). I also created a sub-global model with seed availability and litter depth (*seed_count* + *ld*), representing foraging availability, affecting small mammal relative activity. I included a null and global model for 7 total models. As shrews are insectivorous and have different habitat requirements than rodents and other small mammals, I modeled shrews separately. I used the same models as those for all

small mammals, except I evaluated support for a sub-global model where I predicted litter depth and abundance of invertebrates (*ld* + *invert_count*) affected shrew relative activity. I included a null and global model for 7 total models of shrew relative activity. I checked the goodness of fit for the global model and ranked models based on AICc weight and examined the significance of effect of covariates within the top-ranked model based on if the 85% confidence intervals of β estimates overlap zero.

RESULTS

I identified 5986 independent detections of animals and identified 41 unique species across 59 sites using AHDriFT. This included 25 mammal species, 10 reptiles, and 6 amphibians. I also observed two additional genera of mammals, but *Peromyscus* and *Sylvilagus* individuals could not be reliably identified to species. Including the coverboards and the third camera that viewed the system, I observed 10,443 independent detections of 46 species across 59 sites, with 4 more mammal species from the third camera and 1 more amphibian species from the coverboards (Table 2.3). Across all cameras I had 224 mice, 34 shrews, and 2 salamanders that could not be identified to species. I had 535 complete unknowns that are not included in either totals.

Across 59 sites, the mean Shannon's index of diversity of small mammals was 1.01 (standard deviation 0.27; range 0.42 – 1.62) (Figure 2.2). The mean Shannon's index of small predators was 0.84 (standard deviation 0.47; range 0 – 1.63) (Figure 2.3). For herpetofauna the mean Shannon's index was 0.33 (standard deviation 0.47; range 0 – 1.39) (Figure 2.4, and for invertebrate order it was 0.90 (standard deviation 0.63; range 0 – 1.91) (Figure 2.5).

Objective One - Small mammals and other taxa diversity

Small predators

In predicting small predator Shannon's index of diversity, I found some model selection uncertainty, with 3 models in the 80% confidence set (small mammal Shannon's index and distance to water, distance to water, and an interaction between the two; Table 2.4). I identified small mammal Shannon's index and the distance to water as the top ranking model of the candidate set to predict small predator Shannon's index. This model carried 50% of the model weight, making it 2.29 times more likely than the next ranking model. The effect size of small mammal Shannon's index was 0.40 (85% confidence interval = 0.11 – 0.69) and the scaled effect size of distance to water was -0.22 (85% confidence interval = -0.30 – -0.14), suggesting there was a positive association between small predator Shannon's index and small mammal Shannon's index (Figure 2.3) and water edge (Figure 2.4). Model residual plots suggested this model fit the data well.

In predicting small predator richness, I found that there was high model selection uncertainty, with 3 models in the 80% confidence set (the global, small mammal richness and distance to water, and distance to water; Table 2.5). The global ranked highest and carried 33% of the model weight, making it 1.20 times more likely than the next ranking model. However, the only significant covariate was percent of shrubs (β coefficient -0.39; 85% confidence interval = -0.70 – -0.09). The effect size of small mammal richness was -0.14 (85% confidence interval = -0.30 – 0.02), the scaled effect size of distance to water was 0.02 (85% confidence interval = -1.00 – 1.03), and the effect size of the interaction of small mammal richness and distance to water was -0.13 (85% confidence interval = -0.29 – 0.04), suggesting there was a negative

association between percent of shrubs covering the ground and small predator richness. Model residual plots suggested this model fit the data well.

Herpetofauna

In predicting herpetofauna Shannon's index of diversity, I found that there was high model selection uncertainty, with 5 models in the 80% confidence set (the null, distance to water, small mammal Shannon's index, small mammal Shannon's index and distance to water, and the percent of shrubs; Table 2.6). I identified the null as the top-ranking model of the candidate set to predict herpetofauna's Shannon index. This model carried 27% of the model weight, making it 1.12 times more likely than the next ranking model. The next ranked model, distance to water, was not significant as the scaled effect size of distance to water was -0.09 (85% confidence interval = -0.17 – 0.00). The results from our candidate set suggest herpetofauna Shannon's index does not differ substantially among sites as a function of the variables we measured.

For herpetofauna richness, I found high model selection uncertainty, with 4 models in the 80% confidence set (distance to water, small mammal richness and distance to water, the null, and percent of shrubs) (Table 2.7). I identified distance to water as the top-ranking model of the candidate set to predict herpetofauna richness. This model carried 45% of the model weight, making it 2.71 times more likely than the next ranking model. The scaled effect size of distance to water was -0.30 (85% confidence interval = -0.51 – -0.09), suggesting there was a positive association between herpetofauna richness and water edge. Model residual plots suggested this model fit the data well

Invertebrates

For invertebrate Shannon's index of diversity (calculated at the order, not species, level), I found high model selection uncertainty, with 4 models in the 80% confidence set (percent of

shrubs, the null, percent of shrubs and small mammal Shannon's index, and small mammal Shannon's index) (Table 2.8). Shrubs covering the ground was the top-ranking model of the candidate set, and it carried 32% of the model weight, making it 1.06 times more likely than the next ranking model. The scaled effect size of percent of shrubs was -0.13 (85% confidence interval = -0.24 – -0.01), suggesting there was a negative association between percent of shrubs and invertebrate Shannon's index. Model residual plots suggested this model fit the data well.

For invertebrate order richness, I found there was high model selection uncertainty, with 4 models in the 80% confidence set (percent of shrubs, the null, the percent of shrub and small mammal richness, and distance to water) (Table 2.9). I identified the percent of shrubs covering the ground as the top-ranking model of the candidate set to predict invertebrate richness. This model carried 42% of the model weight, making it 2.04 times more likely than the next ranking model. The scaled effect size of the percent of shrubs was -0.51 (85% confidence interval = -0.90 – -0.13), suggesting there was a negative association between percent of shrubs covering the ground and invertebrate richness. Model residual plots suggested this model fit the data well.

Objective Two - Small mammals and other taxa relative activity

Small predators

For the GLMs predicting small predator relative activity, I found high model selection uncertainty, with 4 models in the 80% confidence set (distance to water, the global, small mammal relative activity and distance to water, and an interaction between small mammal relative activity and distance to water) (Table 2.10). I identified distance to water as the top-ranking model of the candidate set to predict small predator relative activity. This model carried 39% of the model weight, making it 1.92 times more likely than the next ranking model. The scaled effect size of distance to water was -0.35 (85% confidence interval = -0.53 – -0.16),

suggesting there was a positive association between small predator relative activity and drainages. Model residual plots suggested this model fit the data well.

Herpetofauna

For herpetofauna relative activity, I found less model selection uncertainty, with 2 models in the 80% confidence set (distance to water, and small mammal relative activity and distance to water) (Table 2.11). I identified distance to water as the top-ranking model of the candidate set to predict herpetofauna relative activity. This model carried 62% of the model weight, making it 3.08 times more likely than the next ranking model. The scaled effect size of distance to water was -0.39 (85% confidence interval = -0.60 – -0.20), suggesting there was a positive association between herpetofauna relative activity and drainages. Model residual plots suggested this model fit the data well.

Invertebrates

For invertebrate abundance, I found high model selection uncertainty, with 5 models in the 80% confidence set (small mammal relative activity and distance to water, small mammal relative activity, the null, an interaction between small mammal relative activity and distance to water, and the percent of shrubs) (Table 2.12). I identified small mammal relative activity and distance to water as the top-ranking model of the candidate set to predict invertebrate relative activity. This model carried 30% of the model weight, making it 1.68 times more likely than the next ranking model. The scaled effect size of distance to water was 0.20 (85% confidence interval = 0.04 – 0.36) and the effect size of small mammal relative activity was -0.003 (85% confidence interval = -0.005 – -0.001), suggesting there was a negative association between invertebrate abundance and drainages and a small negative association between invertebrate

abundance and small mammal relative activity. Model residual plots suggested this model fit the data well.

Objective Three - Small mammals and microhabitat conditions

For the N-mixture models of all small mammal relative activity, the goodness of fit score (c-hat) for the global detection model was 2.96, so I used QAIC to rank my candidate detection models and inflated my standard error for the 85% confidence interval. I found no model selection uncertainty, the top ranked detection model for small mammals was the global, which carried 99% of the model weight (Supplemental Material Table B2.2). All the detection covariates in the global model were significant. Detection was positively associated with the number of days the camera was knocked over (scaled β-coefficient = 0.08; 85% confidence interval = 0.03 – 0.12) and the interaction of temperature and the days the camera was knocked over (scaled β-coefficient = 0.15; 85% confidence interval = 0.10 – 0.19). Detection was negatively associated with slope (scaled β-coefficient = -0.09; 85% confidence interval = -0.15 – -0.04) and temperature (scaled β-coefficient = 0.16; 85% confidence interval = -0.23 – -0.08). The effect of summer (scaled β-coefficient = -0.85; 85% confidence interval = -0.97 – -0.73) and winter (scaled β-coefficient = -0.77; 85% confidence interval = -0.92 – -0.63) on detection compared to autumn were negative.

I fit N-mixture models for small mammal relative activity using the global detection model. The goodness of fit score (c-hat) for the global relative activity model was 3.49, so I used QAIC to rank my candidate models and inflated my standard error for the 85% confidence interval. I found some model selection uncertainty, with 3 models in the confidence set (total invertebrates, total seeds, and the global) (Supplemental Material Table B2.3). However, none of the effect sizes for the covariates in the models were significant, with all the 85% confidence

intervals crossing zero. I identified the total invertebrates as the top ranked relative activity model for small mammals. This model carried 50% of the model weight, making it 2.66 times more likely than the next ranking model. The scaled effect of total invertebrates on small mammal relative activity was -0.11 (85% confidence interval = -0.32 – 0.09). This suggests that small mammal relative activity is non-significantly negatively associated with invertebrates.

For the N-mixture models of shrew relative activity, the goodness of fit score (c-hat) for the global detection model was 2.01, so I used QAIC to rank my candidate detection models and inflated my standard error for the 85% confidence interval. I found no model selection uncertainty, the top ranked detection model for shrews was the global, which carried 88% of the model weight, making it 7.36 times more likely than the next ranked detection model (Supplemental Material Table B2.4). However, not all detection covariates in the global model were significant. Temperature (scaled β-coefficient = -0.78; 85% confidence interval = -1.22 – -0.33), the number of days the camera was knocked over (scaled β-coefficient = -0.37; 85% confidence interval = -0.69 – -0.06), *Peromyscus* relative activity index (scaled β-coefficient = 0.59; 85% confidence interval = 0.36 – 0.82), and the effect of winter compared to the reference category, autumn, (scaled β-coefficient = -1.31; 85% confidence interval = -2.07 – -0.55) were significant. Slope (scaled β-coefficient = -0.30; 85% confidence interval = -0.74 – 0.13), the interaction effect of temperature and the number of days the camera was knocked over (scaled β-coefficient = 0.20; 85% confidence interval = -0.11 – 0.52), and the effect of summer compared to autumn (scaled β-coefficient = -0.28; 85% confidence interval = -0.41 – 0.96) were not significant. This suggests higher detection probability of shrews in winter compared to autumn, that sites that more effectively detected *Peromyscus* also more effectively detected shrews, and

that shrew detection was negatively associated with temperature and the number of days the camera was knocked over.

I fit N-mixture models for shrew relative activity using the global detection model. The goodness of fit score (c-hat) for the global relative activity model was 2.02, so I used QAIC to rank my candidate models and inflated my standard error for the 85% confidence interval. I found high model selection uncertainty, with 4 models in the confidence set (the null, total invertebrates, litter depth, and distance to water) (Supplemental Material Table B2.5). However, none of the effect sizes for the covariates in the confidence set were significant, with all the 85% confidence intervals crossing zero. I identified the null as the top ranked relative activity model. This model carried 39% of the model weight, making it 1.47 times more likely than the next ranking model. This suggests that shrew relative activity was constant across the study sites, or that we did not include an explanatory covariate for shrew relative activity in the model candidate set.

DISCUSSION

Despite documenting a relatively large number of small mammal species in this study, I found limited support for small mammal communities being good sentinels for other taxa in the southern Blue Ridge. Small mammal diversity and relative activity were not strong predictors of diversity and relative activity of invertebrate and herpetofauna, which, similar to previous studies, were better predicted by habitat characteristics than species associations (Hawkins & Porter, 2003). By contrast I did observe support for associations between small mammal diversity and small predator diversity, supporting the hypothesis that diverse small mammal communities can be indicators of diverse predator communities. Collectively these results suggest that use of small mammals as sentinel species in this ecosystem is limited to certain taxa,

and similar to past global reviews on linkages between mammals and other taxa, these relationships do not ubiquitously apply across different regions and spatial scales (Wolters et al., 2006).

Evidence of a positive association between small mammal diversity and small predator diversity, but not relative activity or richness, supports the hypothesis that diverse prey communities allow for niche differentiation of predators (Dammhahn et al., 2013; Fox, 2004). Experimental studies have shown that prey diversity and composition, not just abundance, supports predator populations as complex prey communities are more reliable, have greater biomass, and provide wider nutritional value (Petchey, 2000). In addition, past studies have found that diverse small mammal communities are associated with the presence of specialized predators (Ayomiposi Ayodele, 2025; Terraube et al., 2011). This could be the case in my study system which was dominated by a single mouse (*Peromyscus*) species, whereby the presence of other small mammal species potentially provided opportunities for other non-*Peromyscus* specialized predators to occupy that area. Diverse small mammal prey communities also allow predators to engage in prey-switching rather than relying on the most abundant prey species in the system (Prokopenko et al., 2025). Alternatively, this association could be explained by bottom-up processes, whereby habitat conditions (e.g., water availability, cover) that led to diverse small mammal communities could also facilitate diverse small predator communities. However, while the sampling scale was sufficient to ensure closure between sites for small mammals, herpetofauna, and invertebrates, the same cannot be said for predator populations. Untangling these competing drivers requires future experimental or manipulative studies.

While I did not observe evidence of associations between small mammals and invertebrates or herpetofauna, relative activities of those taxa were associated with habitat

characteristics, particularly moisture availability. Distance to water was a top predictor in model candidate sets for relative activity for all the taxa I studied, and also predicted small predator and herpetofauna diversity. I found invertebrates to be more diverse and abundant in drier, less shrubby communities. Many of the invertebrates in the soil and leaf litter are larvae, and excessively wet soils can reduce survival of larval insects (Frouz, 1994). Apart from this relationship, all other associations I found between taxa diversity or relative activity and close proximity to water were positive. The negative effect of distance to water is supported by other studies that reported drainages provide cover and water availability for herpetofauna and small predator species (Bateman & Merritt, 2020; Eng & Jachowski, 2019b). Also worth noting is that diversity for all the studied taxa was low. This may have made it especially difficult to find a relationship between small mammals and other taxa, because these communities may already be imperiled.

Failure to find an association between small mammals and herpetofauna could be due to my study design and sampling strategy. First, I based site selection on where I could install AHDriFT systems and where I could optimize small mammal detections, not where I expected to find the most herpetofauna. While AHDriFT was effective in both this study and others for surveying reptiles (Amber, Jr, et al., 2021; Amber, Lipps, et al., 2021; Martin et al., 2017; Proctor, n.d.), it was less reliable detecting salamanders. Salamanders are difficult to detect with motion trigger cameras because they are so small, and as such camera traps targeting them are more successful when triggered at set time intervals instead (Paszkowski, 2011). I also had a small number of observations and limited diversity of salamanders under the coverboards. I only observed one additional amphibian species (Northern Dusky Salamander, *Desmognathus fuscus*) under the coverboards that was not captured on the cameras, and 2 species were unique to the

AHDriFT cameras and not found under coverboards during the study (Black-chinned red salamander, *Pseudotriton ruber* and eastern American toad, *Anaxyrus americanus*). This could be because I used plywood, which a review of salamander coverboard studies reported yields significantly fewer detections than other wood materials due to its desiccating properties (Hesed, 2012). I recommend that future studies that compare the diversity and relative activity of small mammal and herpetofauna supplement camera data with other herpetofauna sampling methods such as natural cover transects, night transects, or pitfalls to optimize salamander observations (Brannon, 2000; Hyde & Simons, 2001).

Failure to find an association between small mammals and invertebrates could be related to relatively low sampling intensity, and from collapsing species to order. I particularly recommend future studies on the relationship between small mammals and invertebrates be conducted because with additional sampling, it would be possible to further investigate if small mammals suppress invertebrate communities. While not statistically significant, the negative association between small mammals in general, and shrews in particular, and invertebrates that we observed is consistent with a grassland study that experimentally found a top-down regulatory effect of small mammals on invertebrates (Churchfield et al., 1991). As such, abundance of small mammals, and highly insectivorous shrews in particular, could not only be poor indicators of invertebrate diversity and abundance, but may reduce availability of prey for other insectivores.

While I did not find support for any of my explanatory models of patterns in small mammal relative activity, this could also be due to the relatively short duration of my study and how I parameterized some of my predictive covariates. Small mammal populations fluctuate year to year, which would not have been captured in my sampling timeframe, where each site was up

for only one year (Korpimäki et al., 1991; Llanos-Guerrero et al., 2024; Norrdahl & Korpimäki, 2002). I recommend more sites and longer-term, multi-year monitoring, especially for shrews, which were relatively infrequently captured compared to other small mammals such as mice and squirrels. It is also possible that the effect of covariates could operate at different spatial or temporal scales than I sampled. For example, small mammal relative activity is reported to be related to the seed availability from the previous year rather than current seed availability (Llanos-Guerrero et al., 2024). The lack of significant results for either seed or invertebrate models could be because I did not have as many seed or invertebrate samples per site as other studies, which completed 18 samples per site for seeds and 30 – 100 samples per site for invertebrates (Churchfield et al., 1991; Duchardt et al., 2021; Llanos-Guerrero et al., 2024). I recommend future studies complete at least 30 samples of invertebrates and seeds per site to model the effect of these covariates on small mammal activity.

In this study I focused survey efforts on small mammal species of conservation concern, but I encourage future studies to examine relationships between small mammals and other taxa and habitat conditions in this region. This study did not survey for raptors, an important small mammal predator, which other studies reported may have top-down and bottom-up trophic interactions with small mammal prey species (Czechowski et al., 2026; Muñoz-Pedreros et al., 2016). The relationships between small mammals and other taxa may also be different in other habitat types, as this study only focused on high elevation forests near drainages and ridgetops. For example, studies in grasslands reported stronger relationships of small mammals to other trophic levels (Duchardt et al., 2021; French et al., 1976). More thorough surveys across additional taxa and habitat types could help clarify when, if ever, small mammals function effectively as sentinels of biodiversity.

Sentinel species for monitoring can be helpful for conservation efforts, but identifying good candidates is difficult. Ideal sentinels are well known, easily surveyed, broadly distributed, sensitive to change, and connected to patterns in other taxa (Pearson, 1994). Novel monitoring techniques like AHDriFT improve knowledge of small mammals and make them easier to monitor, potentially increasing their value as sentinels (Amber, Lipps, et al., 2021; Martin et al., 2017; White et al., 2023). However, despite that small mammals are centrally located in the trophic web of the taxa I studied in this system, my study found that other factors like habitat conditions had stronger associations to other trophic levels than either small mammal diversity or relative activity. This matches a pattern observed in past reviews that suggest using taxa as indicators has limited efficacy, especially across multiple groups (Ricketts et al., 1999). As land managers continue to work to conserve and protect habitat within the southern Blue Ridge, this study suggests that small mammals alone may not capture the full complexity of this highly biodiverse ecosystem, highlighting the need to engage in multi-taxa monitoring.

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TABLES

Table 2.1: Generalized linear models and model predictions I tested for my first and second objectives, to investigate the extent small mammal diversity and relative activity are good indicators of diversity and relative activity of other trophic levels (invertebrates, herpetofauna, small predators).

Hypothesis	**Model**	**Prediction**
Other taxa metric increases with water availability	$Y = \beta0 + \beta1$(distance to water)	$\beta1 < 0$
Other taxa metric increases with the percent ground cover by shrubs	$Y = \beta0 + \beta1$(percent ground cover by shrubs)	$\beta1 > 0$
Other taxa metric increases with the small mammal metric	$Y = \beta0 + \beta1$(small mammal metric)	$\beta1 > 0$
Other taxa metric increases with water availability and the small mammal metric	$Y = \beta0 + \beta1$(distance to water) + $\beta2$(small mammal metric)	$\beta1 < 0, \beta2 > 0$
Other taxa metric increases with the percent ground cover by shrubs and the small mammal metric	$Y = \beta0 + \beta1$(percent ground cover by shrubs) + $\beta2$(small mammal metric)	$\beta1 > 0, \beta2 > 0$
Other taxa metric increases with the small mammal metric, but strength of the effect varies by water availability	$Y = \beta0 + \beta1$(small mammal metric) * $\beta2$(distance to water) + $\beta3$(small mammal metric) + $\beta4$(distance to water)	$\beta1 < 0, \beta2 > 0, \beta3 < 0, \beta4 < 0$
Global	$Y = \beta0 + \beta1$(small mammal metric) * $\beta2$(distance to water) + $\beta3$(small mammal metric) + $\beta4$(distance to water) + $\beta5$(percent ground cover by shrubs)	$\beta1 < 0, \beta2 > 0, \beta3 < 0, \beta4 < 0, \beta5 > 0$
Null – Other taxa metric is constant	$Y = \beta0$	

Table 2.2: Hypotheses I tested for the third objective, to investigate how small mammal relative activity (across all species) and shrew activity is associated with available forage in the southern Blue Ridge and the corresponding N-mixture model and model prediction for each.

Hypothesis	**Model**	**Prediction**
Small mammal relative activity increases with water availability	$\lambda = \beta0 + \beta1(dw) + p$	$\beta1 < 0$
Small mammal relative activity increases with food availability (seeds)	$\lambda = \beta0 + \beta1(seed\_count) + p$	$\beta1 > 0$
Small mammal relative activity increases with food availability (invertebrates)	$\lambda = \beta0 + \beta1(total\_invert) + p$	$\beta1 > 0$
Small mammal relative activity increases with litter depth	$\lambda = \beta0 + \beta1(ld) + p$	$\beta1 > 0$
Small mammal relative activity increases with foraging habitat	$\lambda = \beta0 + \beta1(seed\_count)(all\ small\ mammals)/\beta1(total\_inverts)(shrews) + \beta2(ld) + p$	$\beta1 > 0, \beta2 > 0$
Global	$\lambda = \beta0 + \beta1(dw) + \beta2(seed\_count) + \beta3(total\_invert) + \beta4(ld) + p$	$\beta1 < 0, \beta2 > 0, \beta3 > 0, \beta4 > 0$
Null – Small mammal relative activity is constant	$\lambda = \beta0 + p$	

Table 2.3: Summary of independent observations by species for the study, includes the AHDriFT cameras, the third camera, and the coverboards, and which taxa of analysis they were used for.

Species	AHDriFT	External Camera	Coverboard	Total	Taxa Grouping for Analysis
Peromyscus spp.	3506	568	0	4074	Small mammal
Eastern Gray Squirrel (Sciurus carolinensis)	881	1371	0	2252	Small mammal
American Black Bear (Ursus americanus)	566	469	0	1035	None
Eastern Chipmunk (Tamias striatus)	357	306	0	663	Small mammal
Nine-banded Armadillo (Dasypus novemcinctus)	59	271	0	330	None
Northern Raccoon (Procyon lotor)	79	225	0	304	Small predator
Eastern Woodrat (Neotoma floridana haematoreia)	108	192	0	300	Small mammal
Virginia Opossum (Didelphis virginiana)	131	153	0	284	Small predator
Mouse	188	37	0	225	Small mammal
Sylvilagus spp.	15	119	0	134	Small mammal
Eastern Spotted Skunk (Spilogale putorius)	30	52	0	82	Small predator
Southern Flying Squirrel (Glaucomys volans)	23	59	0	82	Small mammal
Five-lined Skink (Eumeces fascitus)	79	0	0	79	Herpetofauna
Coyote (Canis latrans)	29	37	0	66	Small predator
Northern Short-tailed Shrew (Blarina brevicauda)	62	0	0	62	Small mammal
Bobcat (Lynx rufus)	15	44	0	59	Small predator
Smoky Shrew (Sorex fumeus)	56	3	0	59	Small mammal
Masked Shrew (Sorex cinereus)	56	0	0	56	Small mammal
White-tailed Deer (Odocoileus virginianus)	22	32	0	54	None
Feral pig/hog/swine (Sus scrofa)	7	46	0	53	None
Shrew	33	1	0	34	Small mammal
Woodland Vole (Microtus pinetorum)	20	0	0	20	Small mammal
Southern Gray-cheeked Salamander (Plethodon metcalfi)	2	0	15	17	Herpetofauna
American Pygmy Shrew (Sorex hoyi)	15	0	0	15	Small mammal
Common Gray Fox (Urocyon cinereoargenteus)	0	12	0	12	Small predator
Long-tailed Weasel (Mustela frenata)	3	7	0	10	Small predator
Striped Skunk (Mephitis mephitis)	1	9	0	10	Small predator
Common Garter Snake (Thamnophis sirtalis)	9	0	0	9	Herpetofauna, Small predator
Rat Snake (Elaphe obsoleta)	8	1	0	9	Herpetofauna, Small predator
Blue Ridge Two-lined Salamander (Eurycea wilderae)	1	0	6	7	Herpetofauna
Copperhead (Agkistrodon contortrix)	7	0	0	7	Herpetofauna, Small predator
Northern Slimy Salamander (Plethodon glutinosus)	5	0	2	7	Herpetofauna
Eastern American Toad (Bufo americanus)	5	0	0	5	Herpetofauna
Racer (Coluber constrictor)	3	1	0	4	Herpetofauna, Small predator
Eastern Box Turtle (Terrapene carolina)	2	0	0	2	Herpetofauna
Hispid Cotton Rat (Sigmodon hispidus)	2	0	0	2	Small mammal
Least Shrew (Cryptotis parva)	2	0	0	2	Small mammal
Red Squirrel (Tamiasciurus hudsonicus)	0	2	0	2	Small mammal
Salamander	2	0	0	2	Herpetofauna
Southern Appalachian Salamander (Plethodon oconaluftee)	1	0	1	2	Herpetofauna
Southern Red-backed Vole (Myodes gapperi)	2	0	0	2	Small mammal
Black-chinned Red Salamander (Pseudotriton ruber)	1	0	0	1	Herpetofauna
Brown Snake (Storeria dekayi)	1	0	0	1	Herpetofauna, Small predator
Corn Snake (Elaphe guttata)	1	0	0	1	Herpetofauna, Small predator
Eastern Harvest Mouse (Reithrodontomys humulis)	1	0	0	1	Small mammal
Green Anole (Anolis carolinensis)	1	0	0	1	Herpetofauna
Northern Dusky Salamander (Desmognathus fuscus)	0	0	1	1	Herpetofauna
Red Fox (Vulpes vulpes)	0	1	0	1	Small predator
Scarlet Kingsnake (Lampropeltis elapsoides)	1	0	0	1	Herpetofauna, Small predator
Woodchuck (Marmota monax)	0	1	0	1	Small mammal
Woodland Jumping Mouse (Napaeozapus insignis)	1	0	0	1	Small mammal

Table 2.4: AICc table of generalized linear models for small predator Shannon's diversity index.

Small Predator Shannon's Index Models AICc

Modnames	K	AICc	Delta_AICc	ModelLik	AICcWt	LL	Cum.Wt
Small Mammal Shannon's + Distance to Water	4.00	68.38	0.00	1.00	0.50	−29.82	0.50
Distance to Water	3.00	70.03	1.65	0.44	0.22	−31.80	0.72
Small Mammal Shannon's * Distance to Water	5.00	70.72	2.34	0.31	0.16	−29.79	0.88
Global	6.00	71.24	2.86	0.24	0.12	−28.81	1.00
Small Mammal Shannon's	3.00	81.48	13.10	0.00	0.00	−37.52	1.00
Null	2.00	81.64	13.26	0.00	0.00	−38.71	1.00
Percent Shrub + Small Mammal Shannon's	4.00	82.32	13.94	0.00	0.00	−36.79	1.00
Percent Shrub	3.00	82.36	13.98	0.00	0.00	−37.96	1.00

Table 2.5: AICc table of generalized linear models for small predator richness.

Small Predator Richness Index Models AICc

Modnames	K	AICc	Delta_AICc	ModelLik	AICcWt	LL	Cum.Wt
Global	6.00	227.93	0.00	1.00	0.33	−107.16	0.33
Small Mammal Richness + Distance to Water	4.00	228.29	0.36	0.83	0.28	−109.77	0.61
Distance to Water	3.00	228.95	1.02	0.60	0.20	−111.26	0.81
Small Mammal Richness * Distance to Water	5.00	229.22	1.29	0.52	0.17	−109.04	0.99
Percent Shrub + Small Mammal Richness	4.00	236.68	8.76	0.01	0.00	−113.97	0.99
Percent Shrub	3.00	237.02	9.10	0.01	0.00	−115.29	1.00
Small Mammal Richness	3.00	237.35	9.42	0.01	0.00	−115.46	1.00
Null	2.00	238.99	11.07	0.00	0.00	−117.39	1.00

Table 2.6: AICc table of generalized linear models for herpetofauna Shannon's diversity index.

Herpetofauna Shannon's Index Models AICc

Modnames	K	AICc	Delta_AICc	ModelLik	AICcWt	LL	Cum.Wt
Null	2.00	81.36	0.00	1.00	0.27	−38.57	0.27
Distance to Water	3.00	81.58	0.22	0.89	0.24	−37.57	0.51
Small Mammal Shannon's	3.00	82.71	1.35	0.51	0.14	−38.14	0.65
Small Mammal Shannon's + Distance to Water	4.00	82.81	1.46	0.48	0.13	−37.04	0.78
Percent Shrub	3.00	83.07	1.72	0.42	0.11	−38.32	0.89
Percent Shrub + Small Mammal Shannon's	4.00	84.54	3.18	0.20	0.05	−37.90	0.95
Small Mammal Shannon's * Distance to Water	5.00	85.20	3.85	0.15	0.04	−37.04	0.99
Global	6.00	87.21	5.85	0.05	0.01	−36.80	1.00

Table 2.7: AICc table of generalized linear models for herpetofauna richness.

Herpetofauna Richness Index Models AICc

Modnames	K	AICc	Delta_AICc	ModelLik	AICcWt	LL	Cum.Wt
Distance to Water	3.00	183.48	0.00	1.00	0.45	−88.52	0.45
Small Mammal Richness + Distance to Water	4.00	185.48	1.99	0.37	0.17	−88.37	0.61
Null	2.00	185.61	2.13	0.35	0.16	−90.70	0.77
Percent Shrub	3.00	187.17	3.69	0.16	0.07	−90.37	0.84
Small Mammal Richness * Distance to Water	5.00	187.66	4.18	0.12	0.06	−88.26	0.90
Small Mammal Richness	3.00	187.75	4.27	0.12	0.05	−90.66	0.95
Global	6.00	189.17	5.69	0.06	0.03	−87.78	0.98
Percent Shrub + Small Mammal Richness	4.00	189.27	5.79	0.06	0.02	−90.26	1.00

Table 2.8: AICc table of generalized linear models for invertebrate Shannon's diversity index.

Invertebrate Shannon's Index Models AICc

Modnames	K	AICc	Delta_AICc	ModelLik	AICcWt	LL	Cum.Wt
Percent Shrub	3.00	116.87	0.00	1.00	0.32	−55.22	0.32
Null	2.00	117.00	0.12	0.94	0.30	−56.39	0.63
Percent Shrub + Small Mammal Shannon's	4.00	119.06	2.19	0.33	0.11	−55.16	0.74
Small Mammal Shannon's	3.00	119.13	2.26	0.32	0.10	−56.35	0.84
Distance to Water	3.00	119.16	2.29	0.32	0.10	−56.36	0.94
Small Mammal Shannon's + Distance to Water	4.00	121.37	4.50	0.11	0.03	−56.32	0.98
Global	6.00	123.47	6.60	0.04	0.01	−54.93	0.99
Small Mammal Shannon's * Distance to Water	5.00	123.65	6.78	0.03	0.01	−56.26	1.00

Table 2.9: AICc table of generalized linear models for invertebrate order richness.

Invertebrate Richness Index Models AICc

Modnames	K	AICc	Delta_AICc	ModelLik	AICcWt	LL	Cum.Wt
Percent Shrub	3.00	256.57	0.00	1.00	0.42	−125.07	0.42
Null	2.00	258.00	1.43	0.49	0.21	−126.89	0.63
Percent Shrub + Small Mammal Richness	4.00	258.61	2.04	0.36	0.15	−124.94	0.78
Distance to Water	3.00	260.18	3.61	0.16	0.07	−126.87	0.85
Small Mammal Richness	3.00	260.21	3.64	0.16	0.07	−126.89	0.92
Global	6.00	261.36	4.79	0.09	0.04	−123.87	0.96
Small Mammal Richness + Distance to Water	4.00	262.48	5.91	0.05	0.02	−126.87	0.98
Small Mammal Richness * Distance to Water	5.00	263.15	6.58	0.04	0.02	−126.01	1.00

Table 2.10: AICc table of generalized linear models for small predator relative activity.

Small Predator Relative Activity Models AICc

Modnames	K	AICc	Delta_AICc	ModelLik	AICcWt	LL	Cum.Wt
Distance to Water	3.00	437.66	0.00	1.00	0.39	−215.61	0.39
Global	6.00	438.96	1.30	0.52	0.20	−212.67	0.59
Small Mammal Total + Distance to Water	4.00	439.91	2.25	0.32	0.13	−215.58	0.72
Small Mammal Total * Distance to Water	5.00	440.37	2.71	0.26	0.10	−214.62	0.82
Percent Shrub	3.00	441.24	3.59	0.17	0.06	−217.40	0.88
Null	2.00	441.72	4.06	0.13	0.05	−218.75	0.93
Percent Shrub + Small Mammal Total	4.00	442.23	4.57	0.10	0.04	−216.74	0.97
Small Mammal Total	3.00	442.90	5.24	0.07	0.03	−218.23	1.00

Table 2.11: AICc table of generalized linear models for herpetofauna relative activity.

Herpetofauna Relative Activity Models AICc

Modnames	K	AICc	Delta_AICc	ModelLik	AICcWt	LL	Cum.Wt
Distance to Water	3.00	243.00	0.00	1.00	0.62	−118.28	0.62
Small Mammal Total + Distance to Water	4.00	245.25	2.25	0.32	0.20	−118.25	0.83
Small Mammal Total * Distance to Water	5.00	246.86	3.86	0.15	0.09	−117.86	0.92
Null	2.00	249.15	6.16	0.05	0.03	−122.47	0.94
Global	6.00	249.32	6.32	0.04	0.03	−117.85	0.97
Small Mammal Total	3.00	250.50	7.50	0.02	0.01	−122.03	0.99
Percent Shrub	3.00	251.37	8.38	0.02	0.01	−122.47	1.00
Percent Shrub + Small Mammal Total	4.00	252.79	9.79	0.01	0.00	−122.02	1.00

Table 2.12: AICc table of generalized linear models for invertebrate abundance.

Invertebrate Abundance Models AICc

Modnames	K	AICc	Delta_AICc	ModelLik	AICcWt	LL	Cum.Wt
Small Mammal Total + Distance to Water	4.00	462.40	0.00	1.00	0.30	−226.83	0.30
Small Mammal Total	3.00	463.44	1.04	0.60	0.18	−228.50	0.49
Null	2.00	464.05	1.65	0.44	0.13	−229.92	0.62
Small Mammal Total * Distance to Water	5.00	464.64	2.24	0.33	0.10	−226.75	0.72
Percent Shrub	3.00	464.80	2.40	0.30	0.09	−229.18	0.81
Percent Shrub + Small Mammal Total	4.00	464.92	2.52	0.28	0.09	−228.09	0.90
Distance to Water	3.00	465.75	3.35	0.19	0.06	−229.66	0.95
Global	6.00	466.12	3.72	0.16	0.05	−226.25	1.00

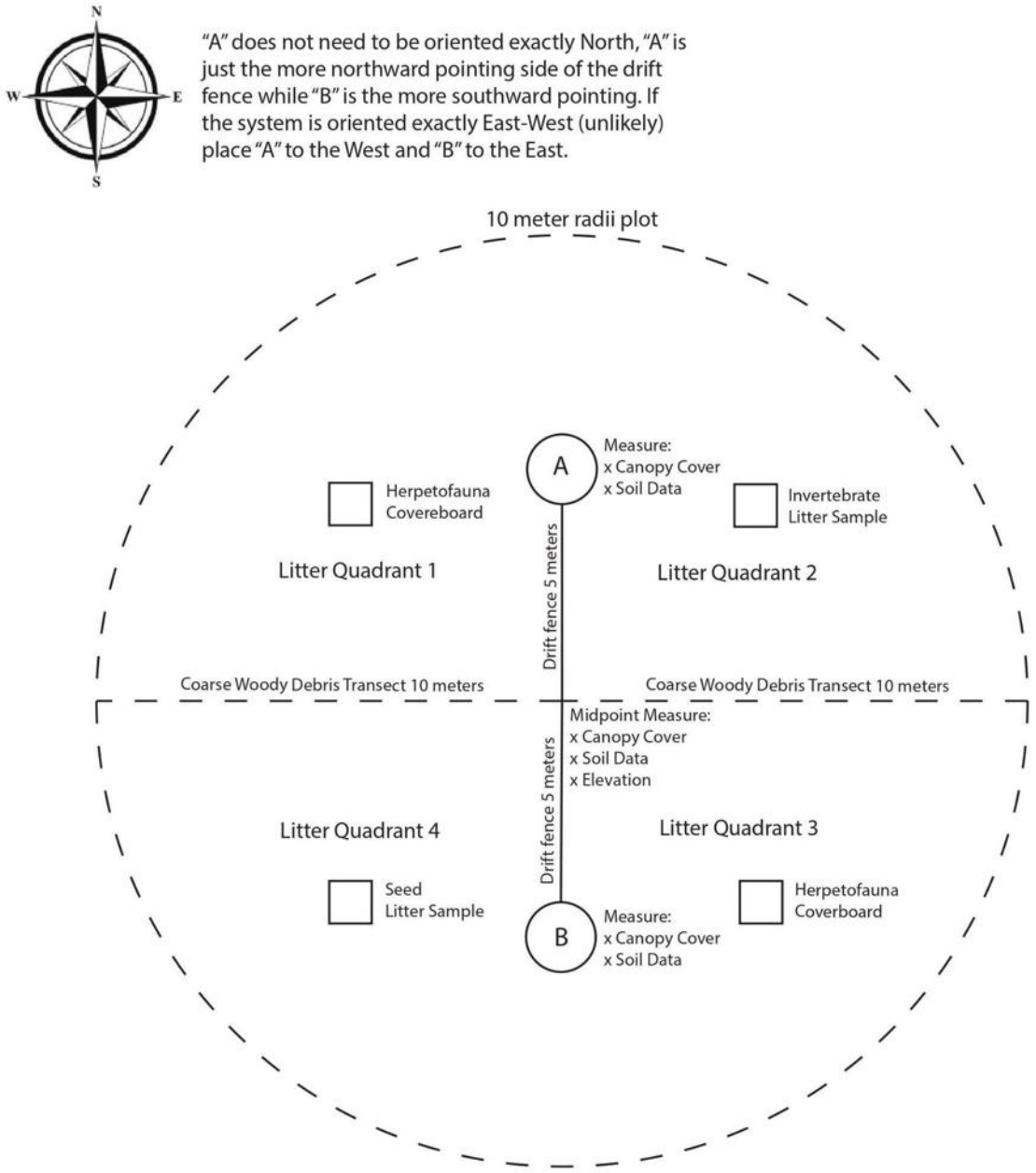
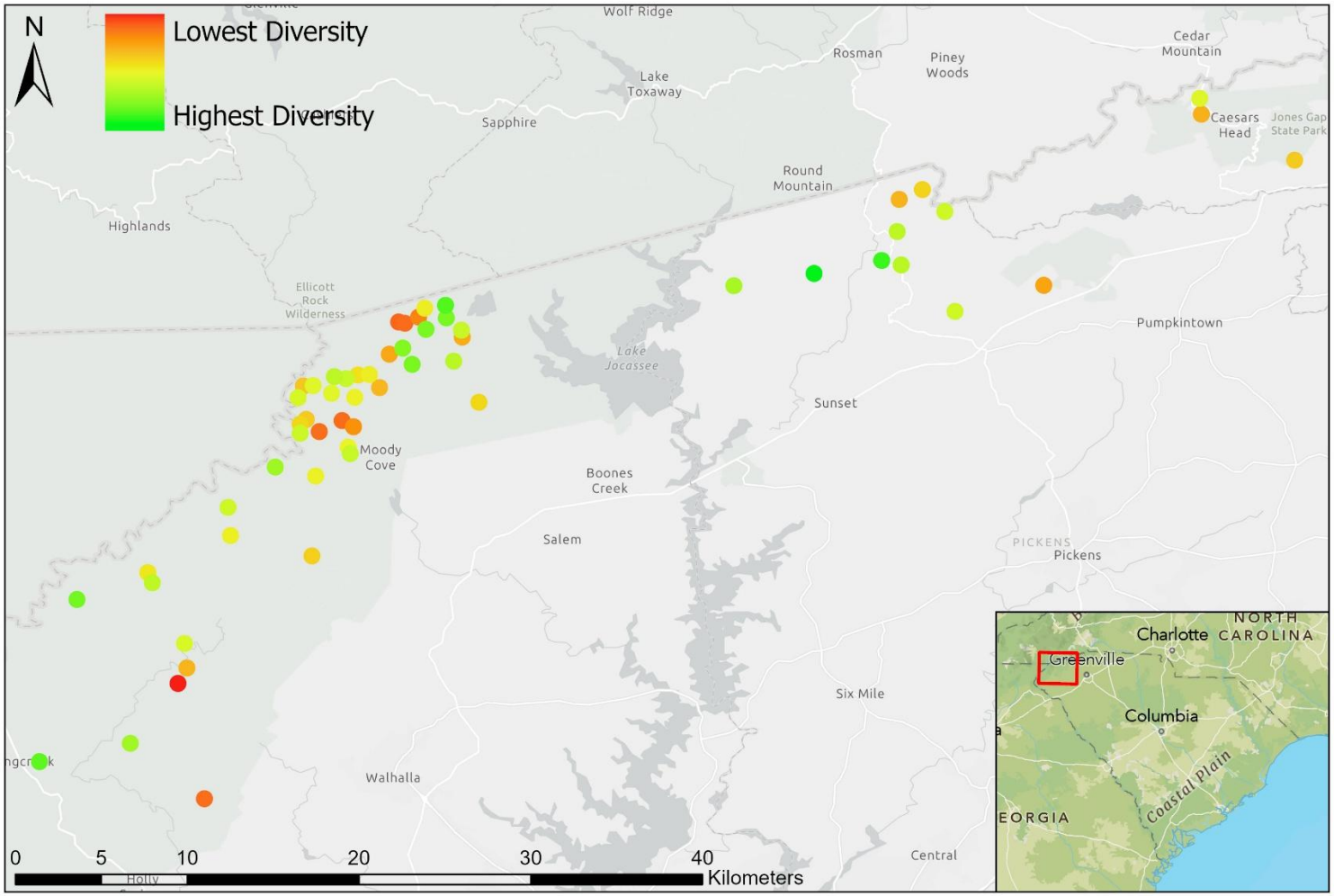
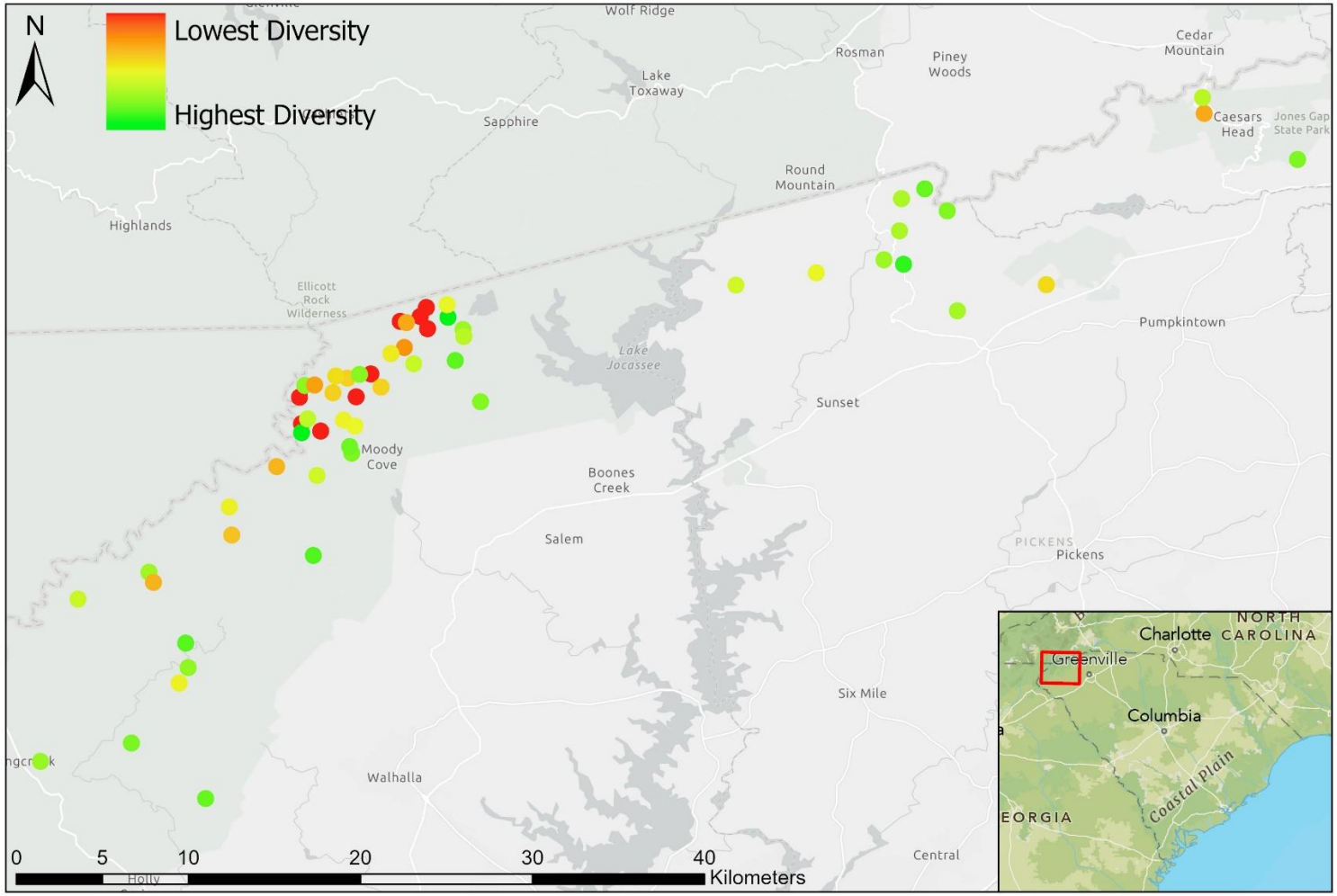


Figure 2.1: Site configuration of AHDriFT system and sampling locations.



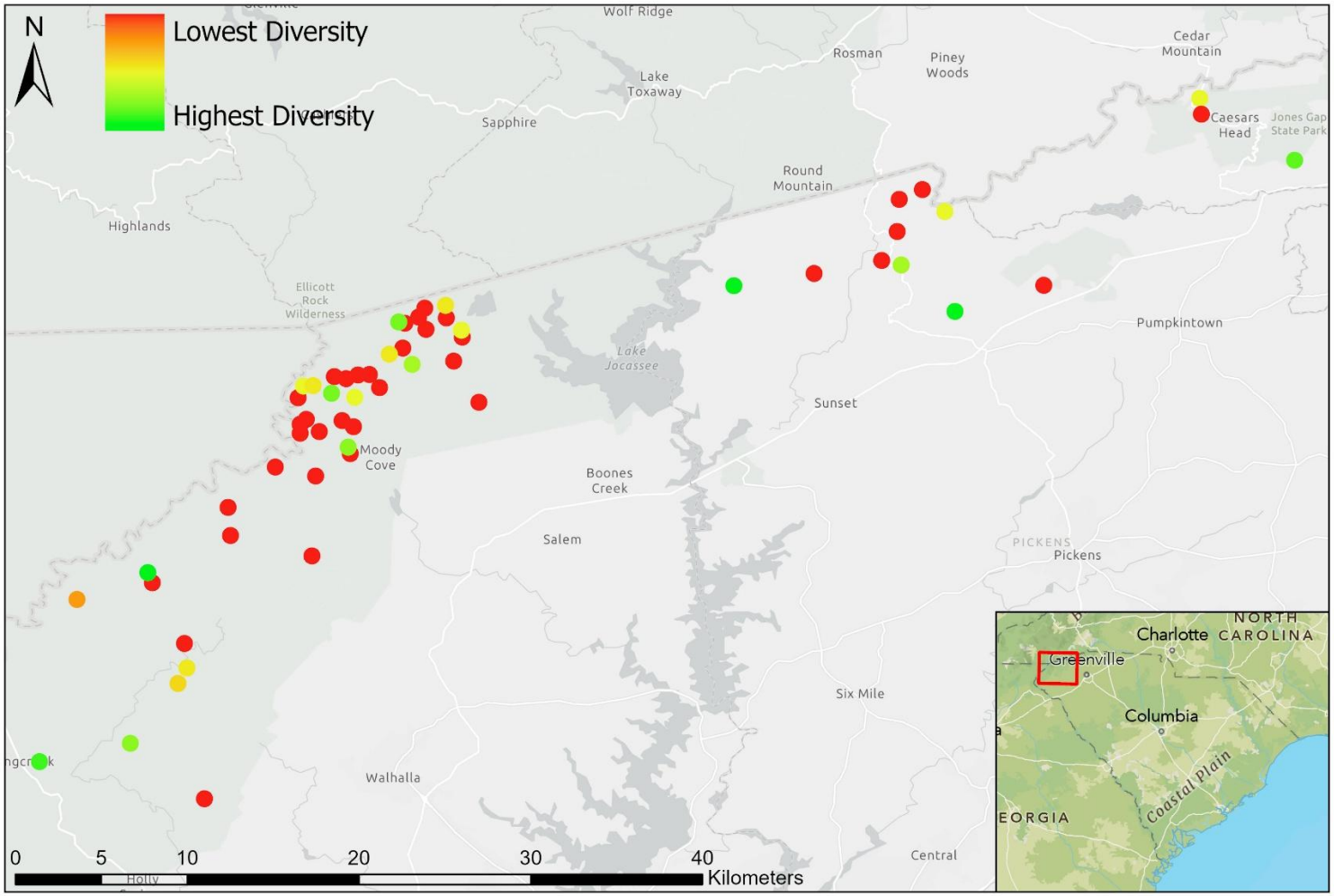
Study Sites by Small Mammal Diversity (Shannon's Index)

Figure 2.2: Map of study sites colored by small mammal Shannon's index



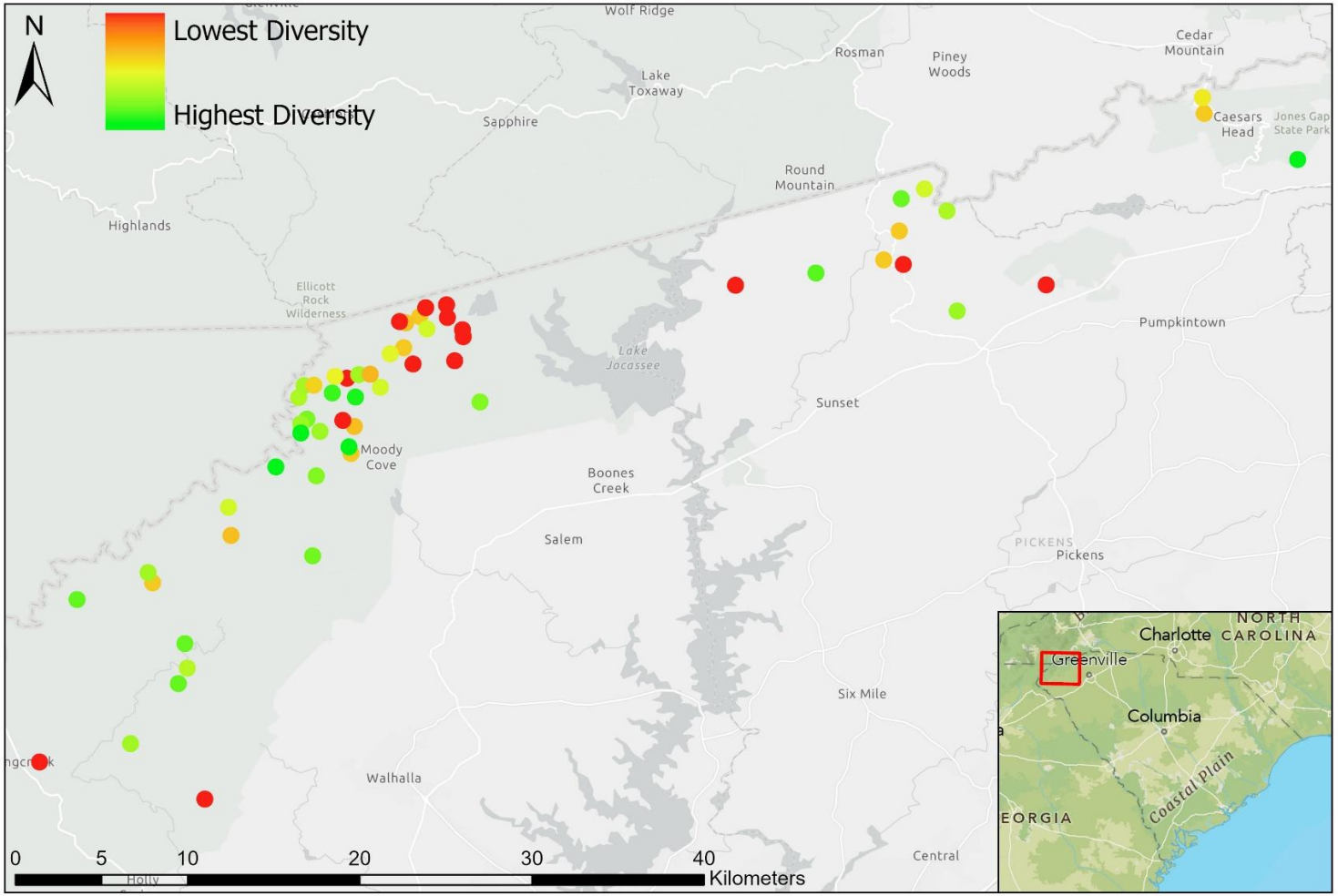
Study Sites by Small Predator Diversity (Shannon's Index)

Figure 2.3: Map of study sites colored by small predator Shannon's index



Study Sites by Herpetofauna Diversity (Shannon's Index)

Figure 2.4: Map of study sites colored by herpetofauna Shannon's index



Study Sites by Invertebrate Order Diversity (Shannon's Index)

Figure 2.5: Map of study sites colored by invertebrate Shannon's index by order

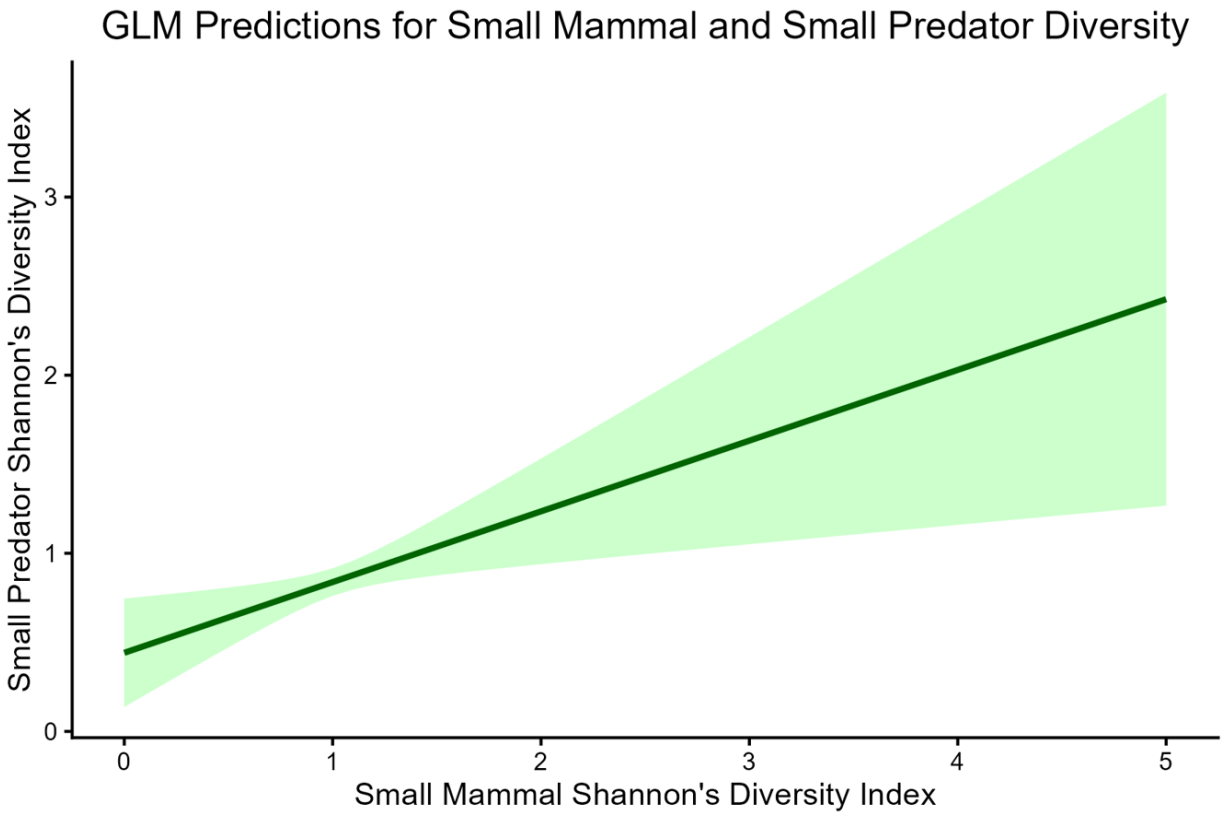


Figure 2.3: Top ranked generalized linear model for small predator Shannon's diversity index. The top model was small mammal Shannon's diversity index + distance to water. Plot shows the effect of small mammal Shannon's diversity index on small predator Shannon's diversity index with the 85% confidence intervals and the effect of distance to water held constant.

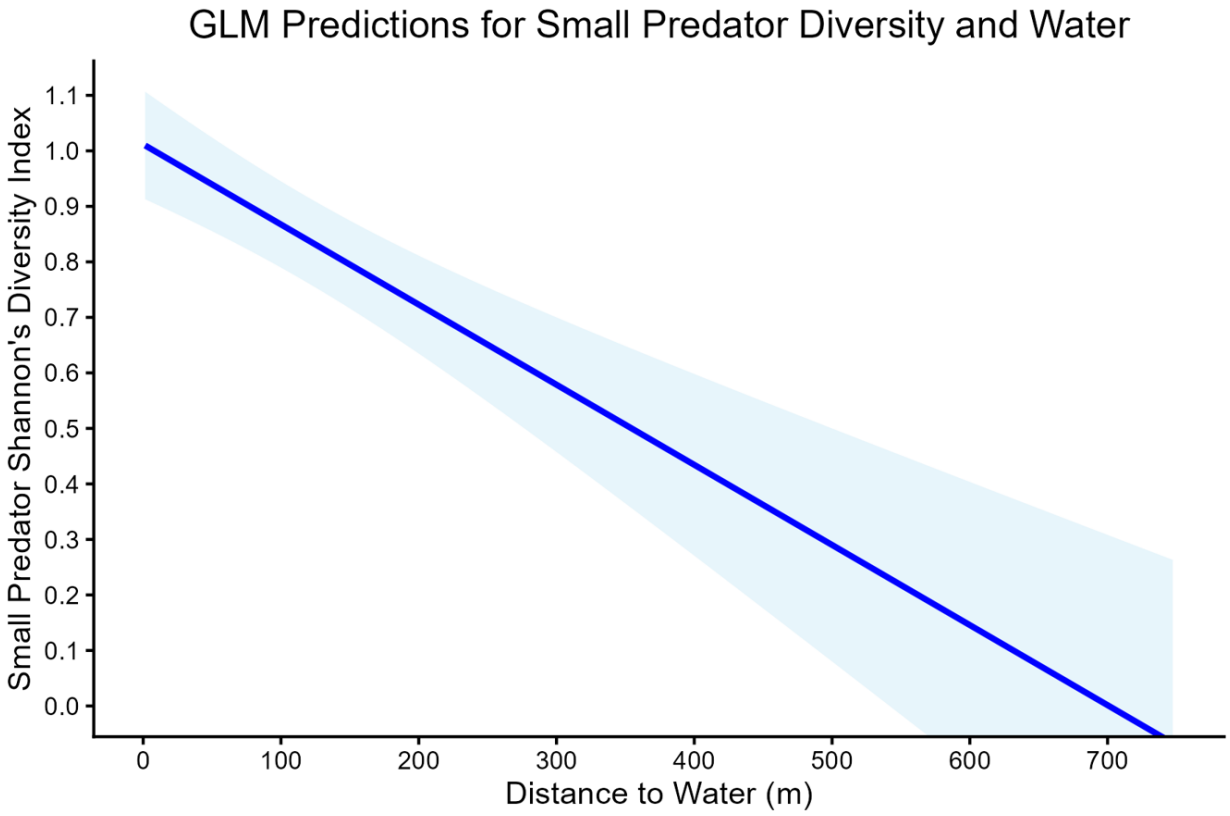


Figure 2.4: Top ranked generalized linear model for small predator Shannon's diversity index. The top model was small mammal Shannon's diversity index + distance to water. Plot shows the effect of distance to water on small predator Shannon's diversity index with the 85% confidence intervals and the effect of small mammal Shannon's diversity index held constant.

APPENDICES

APPENDIX A

Supplemental Material for Chapter 1

Supplemental Material Table A1.1: Summary of prior knowledge of South Carolina 2015 State Wildlife Action Plan listed target species

	Carolina Red-backed vole; *Clethrionomys gapperi carolinensis*	**Eastern Spotted Skunk; *Spilogale putorius***	**Eastern Woodrat; *Neotoma floridana haematoreia***	**Masked Shrew; *Sorex cinereus***	**Pygmy Shrew; *Sorex hoyi***	**Woodland Jumping Mouse; *Napaeozapus insignis***
Habitat	High elevation in SC (above 610 meters to 915 meters) on sky islands, mesic mixed forests, mesic deciduous hardwood, and high altitude coniferous forests and beech gaps, CWD, rotting logs, moss covered rocks, exposed crevices, mountain balds, Rhododendron thickets, bogs. (South Carolina Department of Natural Resources, 2020)	Woods and brush, near farmyards and old fields; dense cover (hedges); CWD; also rock outcrops, barns (South Carolina Department of Natural Resources, 2020)	Habitat generalist; dry or mesic deciduous or mixed deciduous and coves, bottomlands, and swamps. Pine heath or early successional., stands, or rock outcrops, boulder fields and cliffs; more habitat types used in the coastal., plains (South Carolina Department of Natural Resources, 2020)	High elevation, mesic mixed forest, mesic deciduous hardwood, dry deciduous forest, and eastern hemlock ravine forests, thick understory, high soil moisture, high organic matter; logs, stumps, rocks, dense leaf litter(South Carolina Department of Natural Resources, 2020) 790-1370m, Northern slopes (Laerm et al., 1995) 610+ meters (Linzey, 2016)	Mesic mixed forest, mesic deciduous hardwood, dry deciduous forest, and eastern hemlock ravine forests; mountains (South Carolina Department of Natural Resources, 2020)	High elevation (426 meters), cool moist habitat; hemlock stands, sheltered cove hardwood stands; rhododendron streamsides; herbaceous cover; *Endogone* spp. and *Glomus* spp.; may not hibernate in the southeast due to higher temperatures (South Carolina Department of Natural Resources, 2020) North-facing slopes (Brannon, 2005)
Distribution	Counties: Oconee and Pickens (South Carolina Department of Natural Resources, 2020); Walhalla Fish Hatchery (Laerm et al., 1995)	Counties: Aiken, Edgefield, Greenville, Oconee, Pickens (South Carolina Department of Natural Resources, 2020);	Counties: York; also in Coastal., Plains and Sandhills, distribution in Midlands and Upper Piedmont not well understood(South Carolina Department of Natural Resources, 2020)	Little information on location and population densities, but only in high elevation sites (South Carolina Department of Natural Resources, 2020); Oconee County (NatureServe, 2024d); Walhalla Fish Hatchery, 790-1370m (Laerm et al., 1995)	Little information on location and population densities (South Carolina Department of Natural Resources, 2020); Pickens County (NatureServe, 2024e); Walhalla Fish Hatchery (Laerm et al., 1995)	Little information on location and population densities, but only in high elevation sites (South Carolina Department of Natural Resources, 2020); Greenville and Pickens County (NatureServe, 2024b)

Last Sighting	Oconee County 1987-2-21 (NatureServe, 2024a); and 1994 near Walhalla Fish Hatchery and Pickens County near Sassafras Mountain (Laerm et al., 1995)	Greenville and Pickens County 2021-5-12 (NatureServe, 2024f)	Greenville and Pickens County 2005-10-26 (NatureServe, 2024c)	Oconee County 1994-4-17 (NatureServe, 2024d) near Walhalla Fish Hatchery (Laerm et al., 1995)	Pickens County 2000-3-12 (NatureServe, 2024e)	Greenville and Pickens County 1980-1-29 (NatureServe, 2024b)
Conservation Challenges	-S2S3 G5 (SC DNR, 2024); land development, hemlock woolly adelgid (Adelges tsugae) (NatureServe, 2024a)	-S2 GNR (SC DNR, 2024); little known about its true status; habitat fragmentation and public persecution (South Carolina Department of Natural Resources, 2020)	-S3S4 G5 (SC DNR, 2024); land development, predator population density increase (NatureServe, 2024c)	-S2, G5 (SC DNR, 2024); land development, predation by domestic cats and dogs, litter, Adelges tsugae (NatureServe 2024)	-S2, G5 (SC DNR, 2024); land development, predation by domestic cats and dogs, litter (NatureServe, 2024e)	-S2S3, G5 (SC DNR, 2024); land development, Adelgas tsugae, and climate change shifting distribution North (NatureServe, 2024b)
Predated by	Bobcats, weasels, foxes, snakes, and birds of prey (Pivorun et al., 2006)	Great horned owls, bobcats, and domestic pets (Pivorun et al., 2006, Linzey, 2016)	Skunks, weasels, foxes, snakes, and birds of prey (Pivorun et al., 2006)	Foxes, weasels, bobcats, snakes, and birds of prey (Pivorun et al., 2006); domestic cats and dogs (NatureServe, 2024d)	Foxes, snakes, and birds of prey (Pivorun et al., 2006)	Weasels, skunks, mink, bobcat, snakes and birds of prey (Pivorun et al., 2006)
Diet	Seeds, nuts, flowers, berries, fungi, roots, fruits, bark, ferns, lichens, and green vegetation. Endogone is a key food source in late summer and fall. Stores shots, roots, nuts and fungi for winter food (Pivorun	Beetles and other insects, rodents and rabbits, worms, bird eggs, and occasionally fruit (Pivorun et al., 2006)	Eats seeds, berries, nuts, fruits, leaves, stems, tubers, mushrooms, and fungi (Pivorun et al., 2006); consumed more forbs (pokeweed) in summer and browse (eastern redcedar) in winter, generally opportunistic forager	Terrestrial., snails, slugs, insects, ants, spiders, worms, dead vertebrates and occasionally plants; high metabolic rate (Pivorun et al., 2006)	Insects, spiders, ants, seeds, berries, and worms (Pivorun et al., 2006, Linzey, 2016)	Eats seeds, roots, fruits, insects, invertebrates, and Endogone fungus (Pivorun et al., 2006)

	et al., 2006) Inverts (Linzey, 2016)		(McMurry et al., 1993)			
Nest Structure	Globular nests of plant and moss under rocks, roots, logs, or stumps (Pivorun et al., 2006)	Dens in underground burrows, beneath farm buildings, in hollow logs, in rock crevices, or in lumber slab piles; will share dens, especially in winter, but normally solitary (Linzey, 2016; Lesmeister et al., 2008)	Nests the size of a football or larger in underground in hollow logs, under stumps, in trees, in the banks of streams, and in old buildings. Also constructs stick and leaf houses that are 3ft in diameter, and creates debris piles with food, sticks, bones, bark, feathers, duns, paper, colored glass, rags, spoons, and other human trash, hence the name packrat (Pivorun et al., 2006) Uses windthrown root masses (Knowles & Burger, 2008); rock crevices with many fissures (Rossell et al., 2008)	3-4” in diameter of dried leaves and grass under logs, roots, or rocks (Pivorun et al., 2006)	Just below the surface under logs, roots, or in burrows (Pivorun et al., 2006, Linzey, 2016)	Nests in brush piles, under fallen logs, or underground (Pivorun et al., 2006)

Supplemental Material Table A1.2: Eastern woodrat detection models QAICc

Eastern Woodrat Detection Models QAICc

Modnames	**K**	**QAICc**	**Delta_QAICc**	**ModelLik**	**QAICcWt**	**Quasi.LL**	**c_hat**	**Cum.Wt**
Temperature	4.00	71.33	0.00	1.00	0.30	−31.29	2.73	0.30
Null	3.00	71.55	0.22	0.90	0.27	−32.56	2.73	0.58
Peromyscus	4.00	73.44	2.11	0.35	0.11	−32.35	2.73	0.68
Temperature + Days Non-functioning	5.00	73.68	2.35	0.31	0.09	−31.27	2.73	0.78
Days Non-functioning	4.00	73.72	2.40	0.30	0.09	−32.49	2.73	0.87
Slope	4.00	73.77	2.44	0.30	0.09	−32.51	2.73	0.96
Season	5.00	75.47	4.14	0.13	0.04	−32.17	2.73	1.00
Global	9.00	81.06	9.73	0.01	0.00	−29.69	2.73	1.00

Supplemental Material Table A1.3: Eastern woodrat occupancy models QAICc

Eastern Woodrat Occupancy Models QAICc

Modnames	K	QAICc	Delta_QAICc	ModelLik	QAICcWt	Quasi.LL	c_hat	Cum.Wt
Canopy Forest Type	5.00	75.64	0.00	1.00	0.26	−32.26	2.53	0.26
Peromyscus	5.00	75.85	0.20	0.90	0.24	−32.36	2.53	0.50
Null	3.00	76.70	1.05	0.59	0.16	−35.13	2.53	0.66
Coarse Woody Debris Transect Data	5.00	77.90	2.26	0.32	0.09	−33.39	2.53	0.74
Litter Depth	5.00	78.25	2.60	0.27	0.07	−33.56	2.53	0.82
Distance to Water	5.00	78.64	2.99	0.22	0.06	−33.75	2.53	0.87
Percent Ground Cover by Shrubs	5.00	78.65	3.00	0.22	0.06	−33.76	2.53	0.93
Percent Ground Cover by Shrubs + Coarse Woody Debris Transect Data	6.00	80.38	4.74	0.09	0.02	−33.38	2.53	0.96
Litter Depth + Percent Ground Cover by Shrubs	6.00	80.68	5.03	0.08	0.02	−33.53	2.53	0.98
Distance to Water + Percent Ground Cover by Shrubs	6.00	81.11	5.46	0.07	0.02	−33.74	2.53	1.00
Global	10.00	84.54	8.89	0.01	0.00	−29.98	2.53	1.00

Supplemental Material Table A1.4: Eastern spotted skunk detection models AICc

Eastern Spotted Skunk Detection Models AICc

Modnames	K	AICc	Delta_AICc	ModelLik	AICcWt	LL	Cum.Wt
Temperature	3.00	210.98	0.00	1.00	0.66	−102.27	0.66
Temperature + Days Non-Functioning	4.00	213.05	2.08	0.35	0.24	−102.16	0.90
Global	8.00	215.94	4.97	0.08	0.06	−98.53	0.95
Season	4.00	216.62	5.64	0.06	0.04	−103.94	0.99
Peromyscus	3.00	222.46	11.48	0.00	0.00	−108.01	1.00
Null	2.00	223.15	12.17	0.00	0.00	−109.47	1.00
Slope	3.00	223.86	12.89	0.00	0.00	−108.71	1.00
Days Non-Functioning	3.00	224.39	13.42	0.00	0.00	−108.98	1.00

Supplemental Material Table A1.5: Eastern spotted skunk occupancy models QAICc

Eastern Spotted Skunk Occupancy Models QAICc

Modnames	K	QAICc	Delta_QAICc	ModelLik	QAICcWt	Quasi.LL	c_hat	Cum.Wt
Distance to Water	5.00	137.14	0.00	1.00	0.45	−63.00	1.45	0.45
Distance to Water + Percent Ground Cover by Shrubs	6.00	137.50	0.36	0.84	0.37	−61.94	1.45	0.82
Global	10.00	139.54	2.40	0.30	0.13	−57.48	1.45	0.96
Peromyscus	5.00	142.35	5.21	0.07	0.03	−65.61	1.45	0.99
Percent Ground Cover by Shrubs	5.00	146.92	9.78	0.01	0.00	−67.89	1.45	0.99
Litter Depth	5.00	147.61	10.47	0.01	0.00	−68.24	1.45	0.99
Litter Depth + Percent Ground Cover by Shrubs	6.00	148.05	10.91	0.00	0.00	−67.22	1.45	1.00
Coarse Woody Debris Transect Data	5.00	148.75	11.61	0.00	0.00	−68.81	1.45	1.00
Canopy Forest Type	5.00	148.82	11.67	0.00	0.00	−68.84	1.45	1.00
Percent Ground Cover by Shrubs + Coarse Woody Debris Transect Data	6.00	149.03	11.89	0.00	0.00	−67.71	1.45	1.00
Null	3.00	154.04	16.90	0.00	0.00	−73.80	1.45	1.00

Supplemental Material Table A1.6: Masked shrew detection models AICc

Masked Shrew Detection Models AICc

Modnames	K	AICc	Delta_AICc	ModelLik	AICcWt	LL	Cum.Wt
Days Non-functioning	3.00	92.23	0.00	1.00	0.33	−42.90	0.33
Null	2.00	92.81	0.58	0.75	0.25	−44.30	0.58
Temperature + Days Non-functioning	4.00	94.51	2.27	0.32	0.11	−42.88	0.69
Slope	3.00	94.83	2.60	0.27	0.09	−44.20	0.78
Peromyscus	3.00	94.85	2.62	0.27	0.09	−44.21	0.86
Temperature	3.00	95.02	2.79	0.25	0.08	−44.29	0.95
Season	4.00	96.13	3.90	0.14	0.05	−43.70	0.99
Global	8.00	100.25	8.01	0.02	0.01	−40.68	1.00

Supplemental Material Table A1.7: Masked shrew occupancy models QAICc

Masked Shrew Occupancy Models QAICc

Modnames	K	QAICc	Delta_QAICc	ModelLik	QAICcWt	Quasi.LL	c_hat	Cum.Wt
Null	3.00	59.81	0.00	1.00	0.34	−26.69	1.66	0.34
Peromyscus RAI	5.00	62.04	2.23	0.33	0.11	−25.45	1.66	0.45
Coarse Woody Debris Transect Data	5.00	62.20	2.40	0.30	0.10	−25.54	1.66	0.55
Distance to Water	5.00	62.31	2.51	0.29	0.10	−25.59	1.66	0.65
Percent Ground Cover by Shrubs	5.00	62.55	2.74	0.25	0.09	−25.71	1.66	0.73
Litter Depth	5.00	62.77	2.96	0.23	0.08	−25.82	1.66	0.81
Canopy Forest Type	5.00	62.78	2.98	0.23	0.08	−25.83	1.66	0.89
Distance to Water + Percent Ground Cover by Shrubs	6.00	63.89	4.08	0.13	0.04	−25.14	1.66	0.93
Percent Ground Cover by Shrubs + Coarse Woody Debris Transect Data	6.00	63.97	4.16	0.12	0.04	−25.18	1.66	0.97
Litter Depth + Percent Ground Cover by Shrubs	6.00	64.93	5.12	0.08	0.03	−25.66	1.66	1.00
Global	10.00	73.42	13.62	0.00	0.00	−24.42	1.66	1.00

Supplemental Material Table A1.8: Pygmy shrew detection models AICc

Pygmy Shrew Detection Models AICc

Modnames	**K**	**AICc**	**Delta_AICc**	**ModelLik**	**AICcWt**	**LL**	**Cum.Wt**
Null	2.00	87.65	0.00	1.00	0.28	−41.72	0.28
Days Non-functioning	3.00	87.94	0.29	0.87	0.24	−40.75	0.52
Slope	3.00	89.01	1.36	0.51	0.14	−41.29	0.66
Peromyscus	3.00	89.33	1.68	0.43	0.12	−41.45	0.78
Temperature	3.00	89.71	2.06	0.36	0.10	−41.64	0.88
Temperature + Days Non-functioning	4.00	90.11	2.46	0.29	0.08	−40.68	0.96
Season	4.00	92.02	4.37	0.11	0.03	−41.64	1.00
Global	8.00	95.97	8.31	0.02	0.00	−38.54	1.00

Supplemental Material Table A1.9: Pygmy shrew occupancy models AICc

Pygmy Shrew Occupancy Models AICc

Modnames	K	AICc	Delta_AICc	ModelLik	AICcWt	LL	Cum.Wt
Peromyscus RAI	3.00	84.15	0.00	1.00	0.44	−38.86	0.44
Distance to Water	3.00	86.28	2.13	0.34	0.15	−39.92	0.59
Canopy Forest Type	3.00	87.00	2.85	0.24	0.11	−40.28	0.69
Null	2.00	87.65	3.50	0.17	0.08	−41.72	0.77
Distance to Water + Percent Ground Cover by Shrubs	4.00	87.73	3.58	0.17	0.07	−39.50	0.84
Litter Depth	3.00	88.53	4.38	0.11	0.05	−41.05	0.89
Coarse Woody Debris Transect Data	3.00	89.08	4.93	0.08	0.04	−41.32	0.93
Percent Ground Cover by Shrubs	3.00	89.63	5.48	0.06	0.03	−41.60	0.96
Litter Depth + Percent Ground Cover by Shrubs	4.00	90.33	6.18	0.05	0.02	−40.80	0.98
Percent Ground Cover by Shrubs + Coarse Woody Debris Transect Data	4.00	90.92	6.77	0.03	0.01	−41.09	0.99
Global	8.00	92.06	7.91	0.02	0.01	−36.59	1.00

APPENDIX B

Supplemental Material for Chapter 2

Supplemental Material Table B2.1: Invertebrate observations from Berlese-Tullgren funnels

Order	Total
Araneae	144
Argasidae	11
Blattodea	12
Diptera	54
Geophylomorpha	6
Glomerida	1
Hemiptera	6
Isopoda	4
Ixodida	14
Lepidoptera	190
Lithobiomorpha	103
Opiliones	3
Orthoptera	1
Polydesmida	55
Pseudoscorpiones	39
Choleoptera	175
Hymenoptera	306
Amphipoda	1
Opisthopora	5
Scolopendromorpha	1
Spirobolida	1
Siphonaptera	1
Trombidiformes	3

Supplemental Material Table B2.2: All small mammal N-Mixture detection models QAICc table

Small Mammal N-Mixture Detection Models QAICc

Modnames	K	QAICc	Delta_QAICc	ModelLik	QAICcWt	Quasi.LL	c_hat	Cum.Wt
Global	10.00	1,911.98	0.00	1.00	1.00	−943.70	2.96	1.00
Season	6.00	1,943.61	31.63	0.00	0.00	−965.00	2.96	1.00
Temperature * Days Non-functioning	7.00	2,201.11	289.12	0.00	0.00	−1,092.46	2.96	1.00
Temperature + Days Non-functioning	6.00	2,209.53	297.55	0.00	0.00	−1,097.96	2.96	1.00
Temperature	5.00	2,296.56	384.58	0.00	0.00	−1,142.71	2.96	1.00
Days Non-functioning	5.00	2,349.86	437.88	0.00	0.00	−1,169.37	2.96	1.00
Slope	5.00	2,433.60	521.62	0.00	0.00	−1,211.23	2.96	1.00
Null	4.00	2,437.99	526.00	0.00	0.00	−1,214.62	2.96	1.00

Supplemental Material Table B2.3: All small mammal N-Mixture models QAICc table

Small Mammal N-Mixture Models QAICc

Modnames	K	AICc	Delta_AICc	ModelLik	AICcWt	LL	Cum.Wt
Total Invertebrates	10.00	5,537.21	0.00	1.00	0.50	−2,756.31	0.50
Total Seeds	10.00	5,539.17	1.96	0.38	0.19	−2,757.29	0.69
Global	13.00	5,539.69	2.48	0.29	0.14	−2,752.80	0.83
Inverts + Seeds	11.00	5,540.24	3.03	0.22	0.11	−2,756.31	0.94
Litter Depth + Total Seeds	11.00	5,541.42	4.21	0.12	0.06	−2,756.90	1.00
Distance to Water	10.00	5,605.90	68.68	0.00	0.00	−2,790.66	1.00
Null	9.00	5,608.38	71.16	0.00	0.00	−2,793.35	1.00
Litter Depth	10.00	5,610.41	73.19	0.00	0.00	−2,792.91	1.00

Supplemental Material Table B2.4: Shrew species N-Mixture detection models QAICc table

Shrew N-Mixture Detection Models QAICc

Modnames	K	AICc	Delta_AICc	ModelLik	AICcWt	LL	Cum.Wt
Global	10.00	589.33	0.00	1.00	0.88	−282.37	0.88
Temperature + Peromyscus RAI	5.00	593.36	4.03	0.13	0.12	−291.11	1.00
Peromyscus RAI	4.00	603.41	14.08	0.00	0.00	−297.33	1.00
Temperature	4.00	635.66	46.33	0.00	0.00	−313.46	1.00
Temperature + Days Non-functioning	5.00	638.03	48.70	0.00	0.00	−313.45	1.00
Temperature * Days Non-functioning	6.00	640.20	50.87	0.00	0.00	−313.29	1.00
Season	5.00	654.10	64.77	0.00	0.00	−321.48	1.00
Slope	4.00	696.12	106.79	0.00	0.00	−343.69	1.00
Null	3.00	697.40	108.07	0.00	0.00	−345.48	1.00
Days Non-functioning	4.00	699.60	110.27	0.00	0.00	−345.43	1.00

Supplemental Material Table B2.5: Shrew species N-Mixture models QAICc table

Shrew N-Mixture Models QAICc

Modnames	K	AICc	Delta_AICc	ModelLik	AICcWt	LL	Cum.Wt
Null	10.00	589.29	0.00	1.00	0.39	−282.35	0.39
Total Invertebrates	11.00	590.06	0.77	0.68	0.27	−281.22	0.66
Litter Depth	11.00	591.66	2.37	0.31	0.12	−282.02	0.78
Distance to Water	11.00	591.86	2.58	0.28	0.11	−282.12	0.88
Litter Depth + Total Invertebrates	12.00	592.11	2.82	0.24	0.10	−280.66	0.98
Global	13.00	595.26	5.97	0.05	0.02	−280.58	1.00