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Impact of Forest Management on Ozark Big-Eared Bat (Corynorhinus townsendii ingens) Prey and Foraging Ecology

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IMPACT OF FOREST MANAGEMENT ON OZARK BIG EARED-BAT (*Corynorhinus townsendii ingens*) PREY AND FORAGING ECOLOGY

By

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Abstract

Ozark big-eared bats (*Corynorhinus townsendii ingens*) are an endangered species of bat found only in Oklahoma and Arkansas. I conducted a study on the impact of forest management treatments, such as prescribed fire and mechanical thinning, in White Rock Mountain Wildlife Management Area, which is home to a maternity colony of Ozark big-eared bats. My goal was to understand how prescribed fire and mechanical thinning influence Ozark big-eared bat prey and foraging, and therefore be able to provide management recommendations for the habitat of this endangered species. My results indicated that bat prey availability was influenced by forest structure, and was higher in areas that had higher percentages of woody and herbaceous groundcover and more open canopies. Prioritizing these areas through a combination of burning and thinning could be beneficial for bat prey and influence foraging ecology of Ozark big-eared bats. I found that bats were using burned areas less than expected and areas that had not been managed more than expected, but were using thinned areas in proportion to their availability. These results indicate that forest management in this area is not having detrimental effects to bat foraging. I conducted a Hot Spot Analysis to determine potential insect prey availability hot spots, but found that these hot spots were not influencing where bats were foraging. This study provided insight to how the forest has responded to disturbance in this area and information to use in further studies relating to foraging ecology and management for this bat colony.

Keywords: Ozark big-eared bat, insect biomass, forest management, foraging ecology, Arkansas, Hot Spot Analysis

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INTRODUCTION

Bats form one of the largest nonhuman aggregations and are the most abundant group of mammals when measured in numbers of individuals (Jones et al. 2009). They exhibit great diversity and comprise 20% of all mammal species (Perry et al. 2018). There are more than 1400 species of bats and around 45 species live in the United States (O'Shea et al. 2003). Sixteen species exist in Arkansas, including the endangered Indiana bat (*Myotis sodalis*), Gray bat (*Myotis grisescens*), Northern long-eared bat (*Myotis septentironalis*) and Ozark big-eared bat (*Corynorhinus townsendii ingens*).

More than 200 bat species are considered threatened (critically endangered, endangered, or vulnerable) by the International Union for the Conservation of Nature (IUCN) (Bat Conservation International 2022). Habitat loss and fragmentation, disease, and human disturbance are three of their biggest threats. Forests are key habitats for bats, and many species depend on forest attributes such as foliage and cavities of mature trees for roosting which are often reduced in a fragmented forest (Lacki et al. 2007). Human interference and disturbance have also had major impacts on many bat species. Bats are disturbed when humans enter roosting caves, especially those that roost near cave entrances (Beacham et al. 2001). Disease also threatens bats since they are social animals and sociality makes the likelihood of spread high between individuals. White-nose syndrome is a disease that has caused massive population declines in bats throughout North America in the last two decades. It is caused by the fungus *Geomyces destructans,* which manifests itself in the form of thin white spores on a bat’s nose, wings, or ears (Bats and White Nose Syndrome) and produces characteristic skin lesions on the wing and other membranes of bats (Blehert et al. 2009, Courtin et al. 2010, Meteyer et al. 2009). These lesions can result in thinning and tearing of the membrane, hindering the bats’ ability to fly and feed (Bats and White Nose Syndrome 2018). Severely infected bats emerge prematurely from hibernation causing starvation from premature use of fat reserves. White Nose Syndrome was first confirmed in Arkansas caves in 2013 (Bats and White Nose Syndrome 2018), and since then, both Northern long-eared bats and tri-colored bats have seen steep population declines (Bats and White Nose Syndrome 2018) due to the disease.

Since bats provide a variety of ecosystem services, declines in bat populations have negative impacts for ecosystems. Bats are pollinators, seed dispersers, bioindicators, crucial in pest control, and important in soil fertility and nutrient distribution as they are highly mobile and use different habitats, and crucial in transferring nutrients within ecosystems (Buchler 1975, Boyles et al. 2011). Bat guano supports a variety of organisms including arthropods, fungi, bacteria, and lichens, which represent different trophic levels (Polis et al. 1997). Caves lack primary productivity, so guano from cave-roosting bats provides essential organic input that supports assemblages of different endemic cave species, such as cave-dwelling salamanders and invertebrates (Kasso and Balakrishnan 2013). Bats are good bioindicators of habitat quality and serve as bioindicators to disturbance and existence of contaminants (Kasso and Balakrishnan 2013). They are sensitive to accumulations of pesticides and other toxins, and changes in bat abundance can reflect changes in arthropod populations (Hutson et al. 2001). Bats are crucial to ecosystems in many ways, and healthy ecosystems are important in providing various regulatory processes that improve human wellbeing (Chivian and Bernstein 2008)

Declines in bat populations could impact humans greatly. Bats forage on an array of food items including insects, nectar, fruits, seeds, frogs, fish, small mammals, and blood and have important roles in arthropod suppression, seed dispersal, pollination, material and nutrient distribution, and prey and predator interactions (Kunz et al. 2011). More than two thirds of the extant bat species are either obligate or facultative insectivores (Kunz et al. 2011). Insectivorous bats can consume more than 25% of their body mass in insects each night (Coutts et al. 1973), and it is estimated that 99% of crop pests are limited by natural ecosystems, of which some fraction is attributed to bats (Fujita and Tuttle 1991). Their predation can have direct effects on herbivore communities and indirect effects on plant communities through density-mediated (consumption) and trait-mediated (behavioral) interactions (Schmitz and Suttle 2001). Most insectivorous bats eat large quantities of Lepidopterans, Coleopterans, Dipterans, Homopterans, and Hemipterans (Ross 1961, Black 1974, Kunz 1974, Kurta and Whitaker 1998, Kunz et al. 1995, Lee and McCracken 2002, Leelapaibul et al. 2005) and consume enormous quantities of insect pests that cost farmers and foresters billions of dollars (Keeley and Tuttle 1999). A study by Kalka et. al. (2008) indicated that removing bats allowed a 65% increase in arthropod populations, and in turn led to an increase of 68% in leaf damage in comparison to controlled areas, verifying how important bats are in pest suppression. Bats are also important in pollination and seed dispersal, as they can cover long distances while foraging. They help maintain the genetic diversity of flowering plants through pollination (Kasso and Balakrishnan 2013), and foraging fruit bats that defecate in flight can scatter far more seeds across cleared areas than birds (Horner et al 1998). Bats are crucial to their ecosystems, and understanding their habitat and foraging habits is crucial to preservation to many bat species.

Fire has a long history in North America. Historically, it has played a significant role in the ecology of forests by creating disturbance regimes that alter species composition and structure (Brose et al. 2001, Delcourt and Delcourt 1998, Waldrop et al. 1992). It is dominant in establishing and maintaining vegetation patterns in the Ozark Mountains (Cutter and Guyette 1994). In the Missouri Ozarks, the closed-canopy oak-hickory forest that predominates is thought to be an artifact introduced by colonists that interrupted natural ecological cycles by suppressing fire and clearing land for agricultural purposes (Cutter and Guyette 1994). Before the area was settled, travelers in the area described a wide variety of ecosystems including prairies, savannas, and oak-pine forests (Schoolcraft 1821). Analyses have shown that intense logging and farming by settlers decreased fuel loads in the area and prevented fires from igniting and spreading (Cutter and Guyette 1994, Guyette and Cutter 1997, Guyette et al. 2006). In the Ozark Mountains, fire suppression has increased tree density and allowed for the accumulation of leaf litter and debris (Lacki et al. 2009). Prescribed burning and timber harvest were proposed by Hannah (1987) as a management strategy for oaks. Forb groundcover has been shown to increase in areas that have been burned and burned and thinned, and grasses have been shown to increase in burned, thinned, and burned and thinned areas (Kinkead et al. 2013). Shrubs and vines were also seen to increase in following thinning and combination of burning and thinning (Kinkead et al. 2013), but less so than other cover types. The use of prescribed fire is increasing in many eastern forests to produce conditions similar to those before fire suppression existed (Brose et al. 1999, Hutchinson et al. 2005, Van Lear et al. 2000, Waldrop et al. 1992), control populations of insect pests (Martin and Mitchell 1980, Miller 1979, Mitchell 1990), and to reduce fuel loads that could contribute to wildfires (Fettig et al. 2007, McCullough et al. 1998, Van Lear et al. 2004).

There are three types of fires: surface fires, which burn through grass, shrubs, fallen limbs and logs, and needled and leaf litter on the soil surface; crown fires, which are ignited by surface fires and burn through the crowns of standing trees; and ground fires, which burn in subsurface organic fuels (McCullough et al. 1998). The intensity of fire is determined largely by fuel accumulation, distribution, and moisture content. Dead trees, logs, slash, needles, leaves, and litter are all sources of fuel for fires, and the distribution and extent of fuels, aspect, topography, and other factors impact fire intensity and behavior (McCullough et al. 1998).

Both insects and fire are critical components to forest ecosystems. Fires impact species composition, nutrient cycling, and many other ecological processes (McCullough et al. 1998). Insects are greatly impacted by fires, either directly or indirectly through altering soil property, overstory or understory vegetation, tree density, or other habitat components (Lyon et al. 1978, Martin and Mitchell 1980, Mitchell 1990). Immediately following fire, insect abundance can decline (Paquin and Coderre 1997, Siemann et al. 1997), but long-term numeric responses vary (Swengel 2001). Terrestrial insect abundance declines during fires and directly after due to loss of food sources and habitat (Swengel 2001). Fire also burns leaf litter which some insects live in and consume. Fire limits vegetative competition, which allows ground layer species richness to increase

(Kline and McClintock 1994, Wilhelm and Masters 1994, McGee et al 1995). A study by Hutchinson et al. (2005) showed that after five years, burning resulted in small scale increases in grasses, summer forbs and seed banking species. Fire can alter the abundance and spatial-temporal continuity of preferred host species for insects. It can also impact nutrient cycling and availability (Ahlgren 1974, Harvey 1994, MacLean and Wein 1977), consequently determining quality of trees and host plants for some plant-feeding insect species (Attiwell 1994, Rundel and Parsons 1980). Fire creates a cycle of population declines for some insects, followed by recovery and growth periods for other insects and vegetation species (Guerra et al. 2019).

Fire intensity, extent, and frequency in forests shapes the spatial distribution of successional stages (Agee 1993), plant species compositions (Ahlgren and Ahlgren 1960), and availability of fallen wood (Harmon et al. 1986), which all influence the abundance and distribution of wildlife (Saab and Powell 2005, Fontaine et al. 2009, Fontaine and Kennedy 2012). Some moths are attracted to fires, and in the hours following fires, insect abundance can decline substantially, but seasonality of fires relative to insect life stages is important when examining effects of fire on some insects, specifically lepidopterans (Gerson and Kelsey 1997, Miller 1979). Since some moths rely on vegetative hosts, they are expected to be more numerous in burned areas. Those that rely on hosts such as mosses and lichen are expected to be less numerous. In a study by Guerra et. al (2019), it was found that the difference in abundance of *H. fucosa* (painted lichen moth) between burned and unburned sites suggested that prescribed fire did not significantly alter the moss and lichen stratum that the moth relies on. There was a higher abundance of moths recorded at prescribed burn sites in the Ozark Mountains which indicates that fires are impacting moth assemblages (Guerra et al. 2019). Many moths prefer early successional, open canopy habitats that are associated with prescribed burns (Grand and Mello 2004, Noske et al. 2008). Without disturbance from prescribed burns, competition for moths may be more intense. Burned sites may be ideal for moths, as they encourage vegetation growth which serves as a resource for food and habitat, in turn decreasing competition among moths (Guerra et al. 2019). The intermediate disturbance caused by prescribed fire may be ideal for supporting higher moth numbers (Guerra et al. 2019), and therefore may benefit Ozark big-eared bats.

Moths are the most eaten prey of Ozark big-eared bats. However, coleopterans, dipterans, hemipterans, and other insects are also frequent prey items of these bats. In a study of insect communities in the Boston Mountains by Tormanen and Garrie (2020), coleopterans of genus *Phyllophaga* were the most frequently captured. These beetles have many host plants, including a variety of forbs and deciduous trees (Sanderson 1944). The results of the study showed that burning increased forbs but not tree density, and it was determined that prescribed burning may be maximizing the abundance of Coleoptera host plants. The biomass measured of coleopterans was greater in thinned stands, which suggests that thinning may be a strong driver of Coleoptera biomass also (Tormanen and Garrie 2020). In another study by Lacki et al. (2009), there was a higher abundance of coleopterans, dipterans, and all insects combined caught in blacklight traps in postfire conditions when compared to pre-fire conditions. This increase of abundance in insect species could be beneficial to both bats and other insectivorous species. Knowing how to manage the forest for optimal bat foraging could benefit Ozark big-eared bats and increase survivability.

Studies have shown that bat activity increases following disturbances, with increased activity attributed to three possible causes: increased foraging habitat, increased abundance of insect prey, and increased quality and quantity of roosting habitat (Buchalski et al. 2013). Fire reduces vegetation in the forest canopy and understory which can obstruct flyways and interrupt echolocation (Buchalski et al. 2013). Post-fire growth of early successional plant species increases terrestrial insect activity (Lacki et al. 2009, Reed 1997, Swengel 2001, Cooper et al. 2003) and shifts community composition. These conditions likely benefit bat foraging (Buchalski et al. 2013). Fire creates dead and dying trees, which are roosting habitats of some bat species. Responses to habitat change vary greatly among bat species. In a study by Buchalski et al. (2013), bat activity was either equivalent or higher in burned areas than in unburned stands, and bat response was either neutral or positive one year after wildfires. This suggests that bats are resilient to fire and supports the view that fire may increase foraging opportunities for forest bat communities (Buchalski et al. 2013). Studies like this one support the perspective that early successional habitats are essential on the landscape for many taxa, and processes like wildfire are instrumental in maintenance of forest bat communities. I wanted to explore if this pattern applies to the Ozark big-eared bat as well.

Timber thinning is used to create healthy forest ecosystems by reducing overcrowding within the forest. For tree-roosting bats, timber thinning could reduce habitat, but also be used as a source of disturbance that increases heterogeneity of the forest ecosystem. Timber harvesting can mitigate increases in natural disturbance when disturbance events are determined by forest structure (Holmes et al. 2008). The preemptive harvest of live trees may lessen wildfire severity and insect outbreaks, therefore providing benefits beyond the market value of timber (Sims 2013). Thinning reduces stress on plants in the forest by reducing competition. This allows for healthier plants and a more resilient environment for forest-dwelling animals such as bats. A study by Law et al. (2018) showed that the effect of logging history on bats was minor for tree-hollow roosting bats. The survival and abundance of these bats showed minor differences between regrowth and unlogged catchments. Since Ozark big-eared bats are cave-dwellers, logging presumably has minor, if any impacts on the bats; however, timber thinning could impact insects, and specifically the host species of bat prey, which could in turn impact the survivability of Ozark big-eared bats. In a study by Tormanen and Garrie (2021), Lepidoptera biomass was lower at thinned stands, which could negatively impact foraging bats or force them to find other suitable foraging grounds. Another important effect of timber harvesting is a decrease in clutter, which affects bat maneuverability. Thinned areas provide volant species with more area to forage. Ozark big-eared bats have a low wing load ratio (Wethington et al. 1996), which allows for maneuverable flight but is not economical for long distance movements (Weyandt et al. 2005). Timber thinning may improve foraging access or allow Ozark big-eared bats to more easily maneuver through the stand to forage. It is important to understand the management that impacts bat prey to correctly manage for Ozark big-eared bats.

The objective of this study was to assess insect biodiversity and abundance (biomass) at Ozark big-eared bat foraging sites and to determine the impact of forest management practices on these measurements. I hypothesized that vegetation structure would be influenced by forest management, and that burned areas would have a higher moth abundance than unburned areas. I expected an overall increase in insect biodiversity and abundance in managed areas because burned and thinned stands have higher botanical diversity, which could be related to insect diversity.

METHODS

Study Sites

I sampled insects in the home range of an Ozark big-eared bat colony. This colony is located in the White Rock Wildlife Management Area in the Boston Mountain Ranger District of the Arkansas Ozarks. I started data collection in late June and ended mid-July 2022, coordinating dates with an Ozark big-eared bat foraging ecology study. I chose insect trap sites within a 5-kilometer radius of the bat colony. I placed 61 traps randomly in areas that I could access but were also at least 50 meters from any roads to ensure that they were not seen or disturbed. I randomized these sites by taking a random number of steps into the forest to place the traps. Forest stands in this area had previously undergone prescribed burning, understory thinning, commercial thinning, regenerative harvest, or a combination of these practices (Figure 1.1). Preliminary field-based searches took place prior to the start date to identify suitable sites. I selected insect trap sites in each of these different management areas so that I could compare influences of these management practices on vegetation and, insect assemblages, and bat prey availability in the area surrounding the bat colony. Sampling sites were >100m apart to ensure minimal overlap in moth assemblages collected among habitats (Dodd et al. 2008). A pilot study was completed in 2021 to test methods and develop working hypotheses for year two of this study.

Insect Biodiversity and Abundance Surveys

At each site I deployed Universal Black Light Traps (Bioquip, Rancho Domingues, CA, USA) with ultraviolet blacklight powered by a 12 V battery nightly for two weeks with a goal of collecting aerial, nocturnal insects. Bioquip light traps use both visible and ultraviolet light, which have been shown to be the most effective at attracting Lepidoptera when compared to other light traps (Belton and Kempster 1963). The lights had a timer (Misol, Jiaxing City, ZJ, China), which was programmed to turn on 30 minutes before sunset and turn off 30 minutes after sunrise. As insects entered the light traps, they were killed by ethyl-acetate-soaked sponges located in jars at the bottom of the trap, as outlined by Guerra et al. (2019). The light traps were hung from shepherds' hooks at a height of 1.5 m, which helps to control for difference in flight elevation among Lepidoptera families (Taylor and Brown 1972, Lewis et al. 1993).

I deployed six black light insect traps per night for a total of 61 trap nights. Many traps failed due to battery and timer failure, so the study took 14 trap nights to complete. Nights with rainfall were avoided to ensure that insects were identifiable and since rainfall is known to decrease the abundance of moths in light traps (Douthwaite 1978). I collected insects from the traps the morning after they were deployed. I placed insects into in paper bags in a drying oven set to 40 degrees Celsius until fully dried and the mass of the insects stayed stable. Once dried, I identified insects to order, as samples were not well preserved to allow for further identification. For analysis, I focused mainly on the prey of the Ozark big-eared bat, which consists of Lepidopterans, Coleopterans and Megalopterans. I separated Lepidopterans into two sub-categories: Macrolepidopterans and Microlepidopterans, following methods of Dodd et al. (2008). Macrolepidopterans were defined as moths with a wingspan greater than 20 mm, and Microlepidopterans had wingspans less than 20 mm. Macrolepidopterans were a focus of my study as they are a primary prey of the Ozark big-eared bat. I measured total insect biomass in grams for each trap site and biomass of each insect order or sub-category to use in further analysis.

Habitat Assessments and Vegetation Surveys

At each of the selected insect trap sites, I conducted habitat composition assessments and vegetation surveys. I conducted vegetation surveys using a 1-meter Daubenmire frame quadrat and estimated percent groundcover of grass and forbs, vine and woody species, and bare ground or dead organic matter. I then categorized these measurements into classes 1 through 6 based on the range of coverage. Class 1 ranged from 0 to 5% coverage, class 2 ranged from 5 to 25% coverage, class 3 ranged from 25 to 50% coverage, class 4 ranged from 50 to 75% coverage, class 5 ranged from 75 to 95% coverage and class 6 ranged from 95 to 100% coverage. I recorded these measurements in each cardinal direction at each insect trap site and averaged the results from the four plots for data analysis. I estimated tree canopy cover using a spherical densiometer. Understory density (visual obscurity) was measured using a density board (Nudds 1977). I recorded the total number of trees with a DBH of at least 10 cm within 10 m of the trap site and used this as a measurement of mature tree density. Any type of known management practices on the plot, such as fire, regenerative harvest, commercial thinning, and understory thinning, or a combination thereof, were recorded.

Environmental variables that could impact insect activity and bat activity were also be measured and include daily average temperature (degrees Fahrenheit), relative humidity (average %), total precipitation (inches), average wind speed (mph), and percent moon illumination. These measures were taken from the nearest Remote Automated Weather Station (RAWS) to each sampling site (Western Regional Climate Center 2022). The nearest RAWS to the sample sites was the Devil's Knob station. This station was located 70km east of the Ozark big-eared bat colony. Moon phase and illumination were recorded for each sampling night according to the United States Naval Observatory (2022).

Data Analysis

Initially, I wanted to determine if the forest management practices impacted the structure of the forest. To accomplish this, I ran Analyses of Variance (ANOVA) on the habitat variables for each of the four forest management categories. To determine if insect biomass differed among the four forest management categories I ran one-way ANOVAs on biomass of different insect orders to test if insect biomass trapped varied among forest management areas. P-values were considered significant at an alpha-level less than 0.05.

I developed model sets using generalized linear models with insect biomass as the response variable. My objective was to predict insect biomass and assess the impact of forest management practices and forest composition on insect assemblages. Potential explanatory environmental variables included temperature, humidity, percent moon illumination, and windspeed. I eliminated rainfall as an explanatory variable as I avoided sampling on nights with rain and there was only one day of trapping that had rainfall. Potential explanatory habitat variables included elevation, average understory density (visual obscurity), canopy openness, number of trees with a DBH greater than 10cm (mature tree density), and three groundcover components: percent of bare ground and dead organic material, percent of vine and woody plants, and percent of grass and forbs. I used Akaike Information Criterion adjusted for small sample sizes (AICc) to rank models (Burnham and Anderson 2004), and a delta AICc less than or equal to 2 was considered competitive for models. I conducted statistical analyses using Program R version 4.1.3 (R Core Team 2022).

RESULTS

Analysis Of Variance Results

I sampled insects at 61 sites over 14 nights from June 20th to July 4th, 2022. I trapped insects at 17 sites that had not undergone any forest management, 26 sites that had undergone prescribed burns, 7 sites that had been thinned, and 11 sites that had undergone a combination of prescribed burns and thinning (Figure 1.1).

I trapped a total of 1,461g insect biomass and identified eleven different insect orders (Table 1.1) among the 61 sites. These insect orders included Lepidoptera, Coleoptera, Megaloptera, Hemiptera, Plecoptera, Diptera, Homoptera, Odonata, Blattodea, Orthoptera, and Hymenoptera. The number of orders identified at each site ranged from 3 insect orders to 7 insect orders. A Kruskal-Wallis test indicated that there was no statistically significant difference in the number of insect orders identified in different management types ($H_3 = 2.1346$, $p = 0.5449$). Each of the four management types (no forest management, prescribed fire only, thinning only, and a combination of thinning and burning) had a median of six insect orders identified each night (Figure 1.2).

One-way ANOVAs comparing impact of forest management on habitat variables revealed that there were no statistically significant differences in the average mature tree density ($F_{3,57} = 2.736$, $p = 0.0518$), average understory density ($F_{3,57} = 0.046$, $p = 0.987$), or mean percentage of vine and woody groundcover ($F_{3,57} = 0.995$, $p = 0.402$) among the four forest management categories. There was a statistically significant difference in canopy openness between at least two forest management types ($F_{3,57} = 4.397$, $p = 0.0075$). Post hoc analyses were conducted using Tukey's HSD Test for multiple comparisons. These tests found that the mean value of canopy openness was significantly different between the combination of fire and thinning group with the prescribed fire group ($p = 0.026$, 95% C.I. = [-7.225, -0.337]) and group that had not undergone management ($p = 0.01$, 95% C.I. = [-8.242, -0.832]). The combination group had a larger average value for canopy openness when compared to the prescribed fire group and the group that had not undergone forest management (Figure 1.3).

There was a statistically significant difference in average percentage of grass or forb groundcover between at least two forest management groups ($F_{3,57} = 10.84$, $p < .001$). A post hot Tukey's HSD test found that the mean value of percent grass or forb groundcover was significantly different between the combination forest management group and the prescribed fire group ($p < 0.001$, 95% C.I. = [-2.475, -0.650]), the combination forest management group and the thinning group ($p < 0.001$, 95% C.I. = [-3.314, -0.861]), and the combination group and the group that had not undergone any management ($p < 0.001$, 95% C.I. = [-2.890, -0.927]). The combination group had a higher average percentage of grass or forb groundcover than all of the other forest management groups (Figure 1.3).

There was a statistically significant difference in average percent of bare or dead organic groundcover between at least two forest management groups ($F_{3,57} = 6.617$, $p < 0.001$). Post hoc analyses were conducted using Tukey's HSD Test for multiple comparisons. These tests found that the mean value of percent bare or dead organic groundcover was significantly different between the combination forest management group and the prescribed fire group ($p = 0.034$, 95% C.I. = [.059, 2.114]), the combination group and the thinning group ($p = 0.004$, 95% C.I. = [.476, 3.238]), and the combination group and the group that had not undergone any management ($p < 0.001$, 95% C.I. = [.586, 2.797]). The combination group had a lower average percentage of bare or dead organic groundcover than all of the other forest management groups (Figure 1.3).

When comparing different forest management strategies, there were no statistically significant differences in the average total biomass ($F_{3,57} = 1.337$, $p = 0.272$), Lepidoptera biomass ($F_{3,57} = 1.193$, $p = 0.321$), Coleoptera biomass ($F_{3,56} = 1.608$, $p = 0.198$), or Megaloptera biomass ($F_{3,54} = 1.173$, $p = 0.329$) trapped among different forest management areas.

Model Selection: Habitat Variables

The best habitat model predicting insect biomass included average understory density ($\beta = -7.13 \pm 4.24$ SE; Figure 1.5), percent vine or woody groundcover ($\beta = 3.58 \pm 2.11$ SE; Figure 1.5) and elevation ($\beta = -0.04 \pm 0.02$ SE; Figure1.5). The only competitive model (ΔAICc = 1.89) was the null model (Table 1.2).

Similar to the model to predict total insect biomass, the best model to predict Lepidopteran biomass retained average understory density ($\beta = -4.06 \pm 2.21$ SE; Figure 1.7), percent vine or woody groundcover ($\beta = 1.82 \pm 1.10$ SE; Figure 1.7), and elevation ($\beta = -0.02 \pm 0.01$ SE; Figure 1.7) as explanatory variables. The only competitive model (ΔAICc = 1.92) was the null model (Table 1.3).

The best model to predict Macrolepidoptera biomass retained percent vine or woody groundcover (β = 0.30 ± 0.17 SE; Figure 1.9), canopy openness (β = 0.07 ± 0.04 SE; Figure 1.9), and mature tree density (β = 0.11 ± 0.06 SE; Figure 1.9) as explanatory variables. There was no competitive model within 2 ΔAICc to predict macrolepidopera biomass (Table 1.4).

Model Selection: Environmental Variables

The best model to predict total insect biomass using only environmental variables retained only temperature (β = 1.67 ± 0.52 SE; Figure 1.6) as an explanatory variable. The only competitive model (ΔAICc = 1.93) retained temperature and percent moon illumination (β = -0.05 ± 0.09 SE; Figure 1.6) as explanatory variables (Table 1.5).

The best model to predict Lepidoptera biomass using only environmental variables retained only temperature (β = 0.88 ± 0.27 SE; Figure 1.8) as an explanatory variable. There was no competitive model within 2 ΔAICc of this model (Table 1.6).

When using environmental variables to predict Macrolepidoptera biomass trapped, the model with the lowest AICc was the null model. There were two competitive models (Table 1.7): the first (ΔAICc = 0.24) retained percent moon illumination (β = 0.01 ± 0.007 SE; Figure 1.10), and the second (ΔAICc = 1.08) retained percent moon illumination and windspeed (β = 0.16 ± 0.14 SE; Figure 1.10).

Model Selection: Environmental and Habitat Variables

When combining environmental and habitat variables, the best model to predict total insect biomass trapped retained explanatory variables temperature (β = 2.06 ± 0.53 SE; Figure 1.6), elevation (β = -0.06 ± 0.02 SE; Figure 1.5), windspeed (β = 1.20 ± 1.64 SE; Figure 1.6), mature tree density (β = 1.02 ± 0.71 SE; Figure 1.5), and percent vine or woody groundcover (β = 3.07 ± 1.90 SE; Figure 1.5). A competitive model (ΔAICc = 0.24) retained temperature, elevation, mature tree density, percent ground or woody groundcover, plus the two other groundcover components: percent grass or forb groundcover (β = 6.84 ± 4.24 SE) and percent bare or dead organic groundcover (β = 6.75 ± 4.14 SE; Table 1.8). A third competitive model (ΔAICc = 1.21) retained windspeed (β = 2.14 ± 1.70 SE; Figure 1.6) in addition to variables in the second competitive model. The most competitive model to predict total insect biomass trapped suggested that insect biomass increased with higher temperature, windspeed, mature tree density, and percent vine and woody groundcover and decreased slightly at a higher elevation.

The overall best model to predict Lepidopteran biomass retained temperature (β = 1.08 ± 0.27 SE; Figure 1.8), elevation (β = -0.03 ± 0.01 SE; Figure 1.7), windspeed (β = 1.07 ± 0.87 SE; Figure 1.8), average understory density (β = -3.50 ± 2.10 SE; Figure 1.7), mature tree density (β = 0.46 ± 0.37 SE; Figure 1.7), and percent vine or woody groundcover (β = 1.83 ± 0.99 SE; Figure 1.7) as variables. A model including these variables plus percent bare or dead organic groundcover (β = 3.85 ± 2.26 SE) and percent grass or forb groundcover (β = 4.04 ± 2.25 SE) was a competitive model (ΔAICc = 1.95; Table 1.9). Lepidopteran biomass trapped was positively associated with percent vine and woody groundcover, temperature, windspeed, and mature tree density. It was negatively associated with increased understory density and elevation.

The best model to predict Macrolepidoptera biomass trapped was the null model (Table 1.10). A competitive model (ΔAICc = 1.49) retained variables windspeed (β = 0.14 ± 0.13 SE; Figure 1.10), percent moon illumination (β = 0.005 ± 0.007 SE; Figure

1.10), mature tree density ($\beta = 0.07 \pm 0.06$ SE; Figure 1.9), and percent vine or woody groundcover ($\beta = 0.29 \pm 0.18$ SE; Figure 1.9). Macrolepidoptera biomass trapped was positively associated with percent vine and woody groundcover, percent moon illumination, mature tree density and windspeed.

DISCUSSION

Many ecosystems rely on periodic disturbances to disrupt the ecosystem's shift between successional rates (Beisner et al. 2003, Scheffer et al. 2001, Sousa 1984). Fire, drought, and grazing are three of the principal disturbances that shape species assemblages in natural forests and grasslands (Belsky 1992, Collins et al. 1998, Hobbs and Huenneke 1992, Roques et al. 2001). In some cases, intermediate fire frequencies increase heterogeneity of trees and insect abundance and diversity (Connell 1978, Sousa 1979). Fire can regulate competition in vegetative communities, control insect pests and diseases, and facilitate fire-dependent species (USDA Forest Service Southern Region 1989). It alters habitat conditions for birds (Finch et al. 1997) and mammals (Keyser and Ford 2006) and can also greatly impact species by altering availability of food sources. Fire intensity is an important factor in how the fire affects the composition of a forest as more intense fires can result in greater tree mortality (Schwilk et al. 2006).

Responses of forest insects to disturbances such as prescribed fire have been studied extensively but have seen variable outcomes. It seems that the timing of prescribed fire is influential when predicting outcome of these fires (Ferrenberg et al. 2006), as is the number of burns that an area has undergone (Hanula and Wade 2003). Ferrenberg et al. (2006) noticed a higher arthropod species richness in late-season burn treatments than in control areas and early-season burn areas, but noticed a lower abundance of arthropods overall in burned areas. Greater diversity of arthropods in burning treatments could be the result of heterogeneity in habitat (Ferrenberg et al. 2006), since vegetation communities respond variably to fire. Prescribed fires that reduce canopy cover after fires typically lead to more herbaceous groundcover (Moretti et al. 2002), which would be beneficial for diversity and abundance of phytophagous insects. I expected that in the area I studied, insect biodiversity would increase in managed areas, but my results are inconclusive. Even though I did not identify insects to the species level, the diversity of insect orders did not differ among treatments. Measurements for biodiversity, such as species richness, could not be calculated from collected data. Further analysis is needed to determine if insect biodiversity in this area has been impacted by prescribed fire and mechanical thinning.

I expected that groundcover, understory, and canopy measurements would differ among the management areas. Results from my study showed that bare ground or dead organic groundcover was lower in sites that had undergone both prescribed burns and thinning; these sites had higher percentages of grass and forb groundcover, and the canopy was more open when compared to sites that had only undergone prescribed fire or sites that had not undergone any forest management. My results were consistent with other studies, which explained that a combination of prescribed burning and mechanical thinning can improve understory vegetation diversity by reducing canopy tree density (Brose and Van Lear 1998). This is important because in turn, it can be beneficial for Lepidoptera species that use grasses and herbaceous vegetation (Hammond and Miller 1998) as host plants. Prescribed fire alone can have a significant effect on canopy cover and herbaceous groundcover, altering species composition by promoting species that require mineral soil (Franklin et al. 2003) and favoring pioneer species. It can also alter abundance of woody composition and seedling density in forest stands and can increase herbaceous species cover (Franklin et al. 2003). Franklin et al. (2003) explained that in upland oak forests, the greatest increases in herbaceous cover were in stands that had been treated with both thinning and prescribed fire. My results are consistent with the literature in showing that prescribed fire and mechanical thinning allow herbaceous groundcover to thrive because of a more open canopy. These practices could be influential for moths as they alter the ecology of host plants.

There have been several studies showed that insects do not always respond the same way to specific forest treatments. In a study by Holloway et al. (1992), insect species richness varied greatly between undisturbed stands and timber harvested stands, and moth diversity in the early stages of regeneration following clearing was much lower, which makes sense if the moths studied relied on woody host plants. Another study by Taki et al. (2010) explained that species richness and abundance of insects increased in thinned stands when compared to stands that were not thinned. These responses to disturbance can vary based on the intensity of the management effects and the type of management that takes place. In temperate systems, changes in forest structure and composition can be profound and long-lasting (Duffy and Meier 1992). Deforestation in forests of the Appalachian Mountains resulted in only half of the original plant species richness and decreased cover, which could be the result of loss of early-succession herbaceous plants as tree canopies close (Duffy and Meier 1992). Changes in stand structure following logging may drive negative responses of select herbivores, such as Lepidoptera, especially those that are hostplant or microhabitat specialists (Summerville and Crist 2002). Ecological niche, such as hostplant specialization, in moths could be a predictor for community changes after logging, as the richness of hostplant or microhabitat specialists is expected to decrease in disturbed stands, while generalists could be unaffected (Holloway 1989). Woody foliage feeders have been found to have significantly higher species richness and abundance in unlogged stands when compared to clear-cuts, but generalist feeders and species with caterpillars that feed on herbaceous vegetation have had no differences observed in species richness and abundance (Summerville and Crist 2002). The immediate effect of logging is a decrease of large woody structure (McIver and Starr 2001), which could drive negative responses of moths that favor woody host plants, but in the long-term, effects on the forest structure could differ. Both timber harvest treatments and a combination of timber harvest and burning treatments have been found to increase cover of woody species, shrubs, and vines (Kinkead et al. 2013). The response of insect groups often depends on the type of logging, intensity, and ecological niche the insect, which could be part of the reason I did not see differences in insect biomass trapped among the different forest management groups I tested. I did not include intensity, time since the disturbance, or type of thinning treatment in my analysis, which could greatly affect the way that insects respond.

Macrolepidopterans are an important part Ozark big-eared bat prey. Open canopies and higher percentages of grass and forb (herbaceous) groundcover were both positively associated predictors, whereas more bare ground or dead organic groundcover was negatively associated with the amount of Macrolepidopteran biomass trapped. In my study, a combination of prescribed burning and mechanical thinning, which resulted in decreased percentage of bare ground and dead organic groundcover, increased grass and forb groundcover, and increased canopy openness, seemed to provide optimal habitat for Macrolepitoptera. My findings are consistent with the results of Kinkead et al. (2013), whose study found that timber harvest and a combination of management, such as prescribed fire and timber harvest, increased the percentage cover of vines, woody species, and shrubs. Since Lepidoptera are known to use grasses, herbaceous vegetation (Hammond and Miller 1998) and woody species (angiosperms) as host plants, these results show that a combination of prescribed burning and mechanical thinning could increase prey biomass in the bats' foraging area. All three biomass measurements that I tested in this study were higher in areas with more vine and woody groundcover and less bare ground, which seems to be occurring in areas that have been both burned and thinned. Areas that have undergone a combination of management could provide a more suitable foraging area for Ozark big-eared bats, as they promote higher amounts of biomass of their prey.

Groundcover, mature tree density, and understory density were the most influential habitat variables to predict both total insect biomass and Lepidoptera biomass. The similarities in these results are likely due to the large percentage of Lepidopterans trapped in this study. More than 51% of all insect biomass trapped was identified as Lepidoptera. Vine and woody groundcover were influential in the overall best models for total insect, Lepidopteran, and Macrolepidopteran biomass trapped. As a predictor, it had a large effect size in each of the three models that included both habitat and environmental variables, and in the models for total biomass and Macrolepidoptera biomass, it had the largest effect size. This is likely because about 90% of the Macrolepidoptera species within a forest ecosystem depend on the presence of angiosperms as their host plants (Hammond and Miller 1998). In the model for Lepidoptera biomass, percent of vine and woody groundcover had the second largest effect size, as understory density was a more important factor. A denser understory reduced the biomass of Lepidoptera that I trapped. In models including only habitat variables, average understory density (visual obscurity) had the largest effect for total insect biomass trapped and Lepidoptera biomass trapped. There have been other studies that noticed decreased Lepidoptera trap catches in a more dense understory. Dix et al. (1979) used pheromone traps to catch carpenter worm moths. They noticed that light and medium understories did not influence capture rate, but a heavy understory reduced moth catches. Decreases biomass that I witnessed may be a result of a decreased attraction radius of the insect trap, which in a study by Hawes et al. (2009), was significantly negatively correlated with understory density. The differences I saw, in biomass trapped due to higher understory densities, may not be due to a lack of insect biomass near the trap site, but a decreased attraction radius of the insect trap due to a thick understory. The thick understory in some of the trapping areas created a trapping bias which most likely decreased the biomass of Lepidopterans that I trapped.

Increased overall insect biomass, Lepidoptera biomass, and Macrolepidoptera biomass were associated with increased vine and woody groundcover and mature tree density, but my results showed that the current forest management practices in this area do not seem to be having an impact on these habitat measurements specifically. A potential reason why I did not see impacts on these measurements from forest management type alone is that I did not include the amount of time since the management took place as a factor. Incorporating the time since a prescribed burn took place or since thinning operations took place could better highlight differences among management areas, as there are short- and long-term differences in how groundcover types respond to treatment (Rossman et al. 2018). It has been shown that treatment effects can be rarely detected early after prescribed burns that occurred 2 to 3 years prior to the study, but common later, 10 to 13 years after the burn (Rossman et al. 2018). Immediately after fire, basal area of some tree species can be reduced, but overall vegetation density of the area can increase prolifically due to sprouting (Elliott et al. 1999). These results may not be consistent in the long-term. Increases in plant species richness are often slow after burns, and short-term assessments may lead to false conclusions (Rossman et al. 2018). Disturbance frequencies can have a large impact on woody species. Decreases in the average time between fires result in reduced time for plants to accumulate seeds and can extirpate slow growing and maturing species (Enright et al 2015). In closed canopy forests, understory vegetation tends to increase with thinning treatments, as they have the lowest understory component prior to treatment (Stephens et al. 2012). Effects from thinning on more open forests may take years to emerge (Stephens et al. 2012), which is something that I did not consider. In addition to time since management as a factor, many other factors can also impact vegetation dynamics after forest management, such as hillslope. Hillslope gradient can have varying effects on species richness and diversity of vegetation (Elliott et al. 1999) because of exposure to sunlight, which I did not consider in my study. These effects would likely be a large influence on insect assemblage that rely on vegetation for food and shelter.

Another reason why forest management in this area may not be directly influencing groundcover variables the way that I expected is that my management categories (burned, thinned, burned and thinned, and no management) may be too broad. Within the thinned areas, there were stands that had undergone patch clearcutting, single-tree selection curs, shelterwood establishment, sanitation cuts, salvage cuts precommercial thinning, and commercial thinning. The methods for these timber harvest techniques vary. Commercial thinning includes intermediate cutting to stimulate growth and development of a residual stand, which increases the yield of merchantable material in future harvests; precommercial thinning includes removing trees that are too small to be used for lumber; salvage cuts are intermediate cuts that remove dead trees or trees in danger of being killed by insects; sanitation cuts remove dead, damaged or susceptible trees that could become a risk to spread diseases; shelterwood establishment removes a significant portion of the canopy to promote seed production; single-tree selection is used to maintain an even-aged stand; and clearcutting removes essentially all live trees from an entire stand to clear strips that can be managed as a new stand after harvest (Powell 2013). The differences in these practices could have altered the forest structure differently since the methods for each thinning type vary greatly, and impacts from these specific thinning practices may not have been identified in my study. Gradient, time since management, and the type of thinning of forest stands most likely influenced forest structure in this area greatly, and I did not include them as potential explanatory variables. These factors that I did not account for likely affect bat prey in this area, and therefore could impact bat foraging.

Environmental variables such as temperature, percent moon illumination, and elevation were influential predictors for both the total insect biomass and Lepidoptera biomass. Higher moon illumination has been shown to increase Lepidoptera and also has been linked to bat activity and foraging increases (Kolkert et al. 2020). Higher temperature is also known to have a strong influence on insect biomass. Flying insect biomass has been shown to increase linearly with temperature (Welti et al. 2022), which is consistent with my findings. My model selection did not indicate that temperature, percent moon illumination, or elevation were influential factors in predicting Macrolepidopteran biomass trapped, and the only environmental variable that seemed to impact Macrolepidopteran biomass trapped was windspeed. In a study by McGeachie (2009), windspeed was associated with decreased light-trap catches of moths. This result is inconsistent with my results pertaining to Macrolepidopterans but consistent with my findings of Lepidopterans overall. I suspect that the inconsistency in these results could be due to how I measured windspeed. For my study, I used windspeed measurements from the nearest RAWS station that reflected the average windspeed in the area for the day. This station was about 70 km from the Ozark big-eared bat colony and was a similar distance from all of my insect trap sites. I did not measure site-specific windspeed, which is likely to differ from the daily average windspeed at a site 70 km from the insect traps, and therefore could be why my results are not consistent with other Lepidoptera studies that might have used site-specific windspeed.

The relationship of insect biomass trapped with most predictor variables included a slope of zero within confidence interval limits. As these relationships between predictor and response variables are not strong, more research may be required for use in management decisions in Ozark big-eared bat foraging areas. There were some limitations within this study that should be taken into account when determining the implications of this data. The forest system as a whole is complex, and the management categories used throughout this study may not truly identify the forest response to management in the study area. In addition to this, sample sites were mostly located close to a road so that many sites could be reached each day. Time and resource limitations limited the randomness of site selection. Future studies would benefit from truly randomizing study sites and sampling areas of the forest that are farther away from forest roads. Trapping bias was evident in this study because of a limited reach of the insect traps used in forest areas with dense understories. This could potentially have impacted the insect biomass trapped and the relationships of insects to habitat and environmental variables throughout this study.

MANAGEMENT IMPLICATIONS

Different insect taxa require different forest conditions to thrive. Production of moth biomass relies on hardwood trees and shrubs (Hammond and Miller 1998), which is consistent with results from my study. Prioritizing higher mature tree density and higher percentages of woody and herbaceous cover could help produce moth biomass in this area, and in turn benefit Ozark big-eared bats. Basal area can be dramatically reduced after burning (Elliot et al. 1999), which, according to my results, could decrease insect biomass and therefore prey availability for Ozark big-eared bats as it would reduce mature tree density. It may be beneficial to alter the frequency of fire by extending the time between burns to allow for slowly maturing species, such as woody species, to thrive in this area, and therefore provide habitat for bat prey, but keep the interval frequent enough to allow for herbaceous cover to thrive as well.

My study indicated that bare ground or dead organic groundcover was negatively associated with total insect biomass, Lepidoptera biomass, and Macrolepidoptera biomass trapped, and that percentages of bare and organic groundcover were lower in forest stands that had been both burned and thinned than they were in all three other management categories. Thinning allows for canopy openness to increase and can promote greater understory plant species and biomass (Ishii et al. 2008), which provides habitat for insects. In this area, there have been a variety of thinning treatments. These treatments have most likely been variable in their effects on understory plant communities, as their methods vary. I did not examine effects of individual treatments, which would likely alter forest management recommendations as it is probable that the results of my study would differ.

Higher understory density (measured visual obscurity) (Nudds 1977) was negatively associated with insect biomass that I trapped. In areas with a thick understory, there is competition between herbaceous plants in the understory and mature overstory trees (Giuggiola et al. 2018). Reducing the thickness of the understory to less than about twenty percent visual obscurity by conducting mechanical thinning could promote more area for herbaceous or vine and woody groundcover, which I found to have a positive correlation with insect biomass. In my study, herbaceous grass and forb groundcover was higher in areas that had been both burned and thinned than all other management areas. This is likely because the canopy is more open, allowing sunlight to reach the forest floor. A more open canopy was found in areas that had been both burned and thinned, which was influential in increasing Macrolepidoptera biomass trapped. Reducing the competition by removing some of the understory vegetation can cause increased performance in overstory trees (Giuggiola et al. 2018), which is important, as I found mature tree density to be positively associated with all three insect biomass measurements that I tested. Prioritizing a more open canopy, such as one that is ten percent or more open through a combination of burning and thinning could provide suitable habitat for bat prey and optimal foraging areas for Ozark big-eared bats.

Forest management types that I evaluated (prescribed fire, thinning, prescribed fire and thinning, and no management) alone did not have a significant impact on insect biomass that I trapped, though impacts from forest management, such as groundcover types, were found to influence bat prey. The relationship between forest management, insect communities, and bats are complex. A combination of prescribed burning and mechanical thinning in this area of the forest could increase insect biomass by providing habitat with higher percentages of herbaceous groundcover, less bare ground, and more mature trees. Based on results of my study, I recommend maintaining a long enough interval between burns to allow for development of a woody understory, but often enough to promote herbaceous cover over bare ground or heavy organic cover.

CHAPTER II:

INFLUENCE OF INSECT ABUNDANCE ON OZARK BIG-EARED BAT FORAGING

INTRODUCTION

The Ozark big-eared bat (*Corynorhinus townsendii ingens*) is a medium-sized bat with distinctly large ears (U. S. Fish and Wildlife Service 2011). It was listed as endangered by the United States Fish and Wildlife Service on November 30, 1979, because of small population size and limited habitat and distribution. This federally listed endangered subspecies now occurs only in eastern Oklahoma and western north-central Arkansas (Lee et al. 2015). The highly local distribution of the Ozark big-eared bat reflects the limited availability of limestone caves and talus slopes that are commonly used as roost sites (Harvey and Barkley 1990, Clark et al. 1997, Wethington et al. 1996). Populations of Ozark big-eared bats declined by the late 1970's due to human disturbances and the species was eventually extirpated from Missouri (Kunz and Martin 1982). In 1995, the population of Ozark big-eared bats was estimated to be just 1,600 to 2,300 bats (Beacham et al. 2001). These nocturnal insectivores specialize on moths (Van Den Bussche 2016), but also consume beetles and other flying insects. Ozark big-eared bats typically forage in edge and forested habitats (U. S. Fish and Wildlife Service 2011). They have been observed gleaning insects from leaves while perched, but most feed during flight, locating insects using echolocation (Beacham et al. 2001). They are obligate cave dwellers (Humphrey and Martin 1982; U.S. Fish and Wildlife Service 1984, 1995), inhabiting caves that are typically located in hardwood forests year-round and returning to the same maternity caves each year (U. S. Fish and Wildlife Service 2011).

Ozark big-eared bats use limestone caves and talus slopes as roost sites (Harvey and Barkley 1990, Clark et al. 1996, Wethington et al. 1996) and larger hibernation and maternity colonies typically use limestone caves with vertical entrances (Clark et al. 1996; Lacki et al. 1994). Females exhibit philopatry to specific maternity caves (United States Fish and Wildlife Service 1984, 1995; Clark 1991; Clark et al. 1997; Weyandt et al.2005), while males disperse further, maintaining gene flow (Lee et al. 2015). Males remain solitary or stay in small groups during the spring and summer months and can roost at about 100 sites (Lee et al. 2015) on cliff faces, rock fissures, or in caves other than maternity roosts (Clark et al 1993). Males and females segregate during the maternity season (Clark 1991), and females return to traditional maternity sites in northeastern Oklahoma or northwestern Arkansas in spring and summer (Prather and Briggler 2002). There are seven known maternity caves and hibernacula in northwestern Arkansas and thirteen in eastern Oklahoma. Maternity colonies are typically located in relatively warm parts of caves (Harvey and Barkley 1990). Ozark big-eared bats typically have not been observed using buildings and other anthropogenic habitats (Harvey and Barkley 1990).

Parturition and lactation impact how female Ozark big-eared bats forage. Shifts in behavior and foraging activity serve their energy needs (Racey and Swift 1985; Swift 1980). The length of foraging sessions may be constrained by a demand to protect, feed, and regulate temperature of offspring (Barclay 1989). In a study done on foraging activity of female Ozark big-eared bats in summer, there was a reduction of nightly visits to caves and increased foraging time during late lactation, allowing females to replenish depleted fat stores when offspring were large enough to provide their own thermoregulation and forage alone (Clark et al. 1993). In this study, it was shown that during early lactation, females forage about three times a night, returning to the roost between each foraging bout. As lactation progresses, females depart the roost at sunset and return just before sunrise the next morning, increasing the total time spent out of the cave (Clark et al. 1993). Median distances from the maternity cave to foraging sites also increases as lactation progresses (Clark et al. 1993) since food competition near the roost increases when young begin foraging (Kunz 1974). It is assumed that young bats do not travel as far until they become skilled fliers, which would increase the likelihood that adult bats forage farther away from the roost to reduce competition with offspring.

Before parturition, bats leave the roost once per night to forage, and once offspring were born and while they were nonvolant, trimodal activity was noted (Clark et al. 2002). Females have high energy demands while lactating (Kurta et al 1990) and emerge from the roost later, relative to sunset, in June and July. The extra time spent at the roost may be spent grooming and nursing young before emerging to forage (Shen and Lee 2000). Once the offspring can fly and forage for themselves, most Ozark big-eared bats leave caves after sunset and do not return until sunrise, which is presumed to represent the decreased need for offspring to nurse (Clark et al. 2002). Most subspecies of C. townsendii are late to emerge from the roost, not leaving until little to no twilight remains (Kunz and Martin 1982), but most Ozark big-eared bats emerge at sunset or shortly thereafter (Clark et al. 2002). A study by Rydell et al. (1996) suggested that emergence of insectivorous bats was determined partly by the timing of insect flights.

Variation in prey selection was related to interspecific difference in emergence times of bats. Differential availability of prey throughout the night impacts the emergence time of insectivorous bats (Clark et al. 2002), as does cloud cover and other elements of brightness (Erkert 1982, Fenton et al. 1977), such as bright moonlight.

Foraging habitat for Ozark big-eared bats includes edge habitat more than expected and forested habitat less than expected, while open habitat is used in proportion to its availability during early and late lactation but was avoided mid-lactation (Clark et al. 1993). Edge habitat may be preferred because it provides cover for bats and their primary prey, moths (Clark 1991). Uncluttered habitat allows for easier feeding for bats but less structural protection from predators (Erkert 1982). When Ozark big-eared bats forage along woodland edges, they benefit from a less-cluttered environment while still having nearby cover and high prey densities (Clark 1991). Trees provide cover and an abundance of moths for bats, but the habitat is more cluttered. Ozark big-eared bats that forage along woodland edges benefit from a less-cluttered environment that still has cover nearby and high prey densities (Clark 1991). Forest habitat is crucial for survival of the Ozark big-eared bat. Threats to forests along with other human-imposed pressures have the potential to a variety of problems for these endangered, forest-dwelling bats.

The biggest threats to Ozark big-eared bats are vandalism, human disturbance, and habitat fragmentation (U. S. Fish and Wildlife Service 2011). Maintaining suitable habitat through cave gating or through other means is crucial for the future of the Ozark big-eared bat, and management for their insect prey could provide a significant benefit in this task.

Understanding foraging ecology of Ozark big-eared bats is crucial to their longevity. By determining where they forage, we may be able to understand factors that are impacting why they are foraging at certain locations, such as insect abundance and forest management practices. The objectives of this study were to determine whether management influences on bat prey affect Ozark big-eared bat foraging ecology and to understand the spatial distribution of insect abundance relative to habitat characteristics near an Ozark big-eared bat maternity colony.

I expect that managed areas will provide Ozark big-eared bats with superior foraging grounds, therefore increasing foraging activity in those areas. I expected that insect abundance will be higher in managed areas and, which will increase foraging in those areas. I hypothesized that bat foraging locations will be near to identified insect hot spots, as prey could be more available to bats in these areas.

METHODS

Study area

This study took place within a 5-kilometer radius of the Bliss Ridge Ozark big-eared bat maternity colony in the White Rock Wildlife Management Area in the Boston Mountain Ranger District of the Arkansas Ozarks (Figure 2.1).

Telemetry and bat foraging

Radio-tagging is a unique tool that allows researchers to track bats by marking bats with a small radio-transmitter (Wilkinson and Bradbury 1988). Using these radio-transmitters for tracking bat locations allow researchers to gain knowledge on bat behavioral ecology and contribute to conservation (Bontadina et al. 1999). Once the radio-transmitter is attached and the bat is released, researchers are able to estimate locations of the animals using azimuth data. These data can then be used to understand the animal's home range and foraging area. Wildlife radio telemetry was used to track the movement of Ozark big-eared bats. Bat movement was tracked for fourteen days to understand foraging activity and home ranges of bats. Both ground-based telemetry and aerial telemetry were used. We captured seven bats using mist nets on June 20th, 2022, and seven bats on June 27th 2022. Mass, forearm length, reproductive status, and general health (e.g., presence of parasites or wing damage) was recorded to assess age structure and health of the population for other studies. Bats were fitted with LB-2x transmitters, obtained from ®Holohil Systems Ltd., between their shoulder blades using permatype cement. Each transmitter weighed 0.01 ounce (0.42g), which did not exceed 5% body mass (Sikes et al. 2011). After processing, bats were held for ten minutes to allow glue to dry and allow bats to recover from stress related to handling (Moore et al. 2017).

Triangulation began at sunset. Radio-telemetry sites were chosen based on known bat locations and were changed, when necessary, in order to achieve the most precise bat locations. Stations were selected both for proximity to previous bat locations and for higher vantage points. Azimuths were compiled for individual bats and entered into the program Location of a Signal (LOAS), which uses several parameters to output location estimates and error polygons. Estimated parameters were based on Maximum Likelihood Estimator (MLE) of available data and were used to determine home range, or the area polygon in which an individual spends 95% of its time within a discrete time period, such as summer foraging. The core-foraging range is an area where an individual spends 50% of its time in a discrete time period. Estimates of home range and core-foraging area were provided to this study by Patrick Moore (Environmental Solutions and Innovations, Inc.)

A chi-squared test for independence was used to determine if bat foraging locations are influenced by forest management within the Ozark big-eared bat colony's home range. I expected that there would be a difference in the proportion of bat foraging locations among the four forest management types, as insect assemblages can be altered by prescribed burns and mechanical thinning.

Modeling Insect Hot Spots

I trapped insects using Universal Black Light Traps with ultraviolet blacklight powered by a 12 V battery nightly for two weeks. I wanted to trap nocturnal, aerial insects that would represent available prey of the bats in this foraging study. I used ethyl acetate as a killing agent in the traps, and dried insects in a drying oven before I identified insects to order. Once separated to order, I measured the biomass (g) of each insect order. For this analysis, I focused on total biomass measurements collected at each site, Lepidoptera (moth) biomass, and Macrolepidoptera (macromoth) biomass, as the bats in this foraging study are moth specialists. I used the biomass measurements taken at each trap site to model hot spots of bat prey availability.

A hot spot analysis is a qualitative assessment instrument that estimates resource-intensity or other indicator areas (Liedtke et al. 2010). The main objective is to identify peaks of a resource, such as available prey. I used the Hot Spot Analysis tool in ArcMap to identify significant spatial clusters of high and low insect abundance. This analysis consists of a null randomization model test of a sample distribution of spatial attributes (Piorkowski and O'Connell 2010). For each weighted data point in a projection, the tool calculates a Getis-ORD Gi* statistic, which is interpreted as a Z-score with an ALPHA = 0.05. The Getis-Ord Gi* statistic is based on attribute similarity along the point and all neighboring points. The test assumes spatially random distribution of the proportional sum of the total attribute value for the entire sample, with unusually high proportions of the attribute sum located at a point or points resulting in high Z-scores and interpreted as spatially significant (Piorkowski and O'Connell 2010). If the measured biomass of a certain location is significantly higher than the study area, then it is marked in red. This tool also identifies cold spots, which are marked in blue. In my study, the location of hot spots for insect biomass means that there is high availability of bat prey.

Foraging Activity and Insect Hot Spots

After defining insect hot spots, I determined whether a bat foraging location was more likely to be near to a hot spot than a random point. I created a random point dataset within the 2022 foraging area using the Create Random Points tool in ArcMap, and analyzed results using a Wilcoxon rank sum test in Program R to compare the distances between bat foraging locations to hot spots with bat foraging locations to random points.

RESULTS

Bat Foraging

During the two-week Ozark big-eared bat foraging study, there were 349 bat locations determined within the 2,072-hectare home range of the studied Ozark big-eared bat colony (Figure 2.1). Of these locations, 141 were within areas that had not undergone any forest management, 160 were within areas that had undergone prescribed fire, 42 were in areas that had undergone timber thinning, and 6 locations were in areas that had undergone both prescribed burns and thinning (Figure 2.2). A chi-square test of independence was performed to examine the relation between bat foraging locations and forest management type. The relationship between these variables was significant, (X2=23.659, df=3, p<.001). The bats in this study foraged in areas that had not undergone management more than expected, and areas that had been burned less than expected (Figure). About 29% of the home range of the bats had not undergone management, and more than 40% of the bat locations were located within this area. Fifty-five percent of the area had undergone prescribed fire, and about 46% of the bat locations were located within this area.

Hot Spot Analysis

I trapped a total of 1,461g of insect biomass at 61 sites. Of this total, 751g were Lepidopterans and 96g were Macrolpidopterans. Total insect biomass trapped varied from 1.5g to 71g, and the average insect biomass trapped per site was 23.9g. Lepidoptera biomass trapped varied from 0.4g to 36g, with an average of 12.3g trapped per site. Macrolepidoptera biomass trapped varied from 0 to 5.8g, with an average of 1.6g trapped per site.

Hot Spots of trapped insect biomass was determined using the Hot Spot Analysis tool in ArcMap. Sixty-one insect biomass measurements were used as input for the Hot Spot Analysis. At the 95% confidence level, there was one total insect biomass hot spot determined, and this site was not within the Ozark big-eared bat home range (Figure 2.3). This site was within an area that had only been burned and had not undergone any timber thinning. This was also true when using Lepidoptera biomass as an input for the hot spot analysis (Figure 2.4). Further analysis was not completed to determine whether foraging bats were more likely to be located near these hot spots, as they were outside the home range and core foraging areas determined for the bats in this colony.

When using Macrolepidopteran biomass as an input, there were 7 total hot spots identified at the 95% confidence level (Figure 5). Three of the hot spots were within areas that had undergone prescribed burns, and four of the hot spots were in areas that had not undergone prescribed fire or timber thinning. There was also one cold spot identified in this analysis. The cold spot was in an area that had undergone prescribed fire. Six of the hot spots were located within the home range of the bats. These six points were used in the analysis to determine if foraging locations were more likely to be near these hot spots or two six randomly generated locations (Figure 2.6). The median distance from a bat foraging location to the nearest Macrolepidoptera hot spot was 1,281.7 meters, and the median distance from a bat foraging location to one of six randomly generated locations was 883.1 meters. A Wilcoxon rank sum test determined that there was a significant difference in the median distance to the nearest hot spot or randomly generated point (W=68,917, p<.001). According to this analysis, bats were more likely to be located near to a randomly generated point than they were to be near a Macrolepidoptera hot spot.

DISCUSSION

Bat Foraging

Of the 349 Ozark big-eared bat foraging detections recorded in this study, only 48 of these locations were in forest stands that had been thinned. The remaining locations were in areas that had either been prescribed fire or had not been burned or thinned, showing that the bats of this maternity colony were not actively looking to forage in thinned areas. This contradicted my assumption that less cluttered areas could provide superior foraging grounds for these bats.

Ozark big-eared bats are moth specialists. Moth diversity and species composition is dependent on diversity of plants (Axmacher et al., 2009). Increased living wood volume increases niche space or food resources with available leaf area as habitat, cover, and food for moths (Delabye et al., 2021), which could be lessened with mechanical forest treatment. There are three moth families that represent a large proportion of the Ozark big-eared bats diet: Erebidae, Noctuidae, and Notodontidae (van den Bussche et al. 2016). All of these moths are considered macromoths, as they typically have a larger wingspan, and according to Dodd et al. (2008), moths with wingspans of 20 mm or greater are part of the subgroup Macrolepidoptera. Moths in the family Erebidae (formerly Arctiidae) are common. Larvae in this group feed on grasses, trees, and shrubs (Borror and White 1998). Noctuid moths vary greatly in size and color, but most have a wingspread of 20 to 40 mm (Borror and White 1998), classifying them as Macrolepidopterans. The larvae of these moths bore into stems and fruits. Notodontids, also Macrolepidopterans, are common moths that have gregarious larvae that feed on trees and shrubs. All of these moths, in some way, require woody species as host plants. In other studies, more than 80% of the abundance of Macrolepidoptera was dependent on angiosperms (Hammond and Miller 1998). Hardwood trees and shrubs are crucial to the production of moth biomass (Hammond and Miller 1998). My results confirmed these trends, as I found higher Lepidoptera and Macrolepidoptera biomass in areas with higher mature tree density and more vine and woody groundcover.

In a previous study on Rafinesque’s big-eared bats (*Corynorhinus rafinesquii),* Erebidae, Noctuidae, and Microlepidopterans were found in greater numbers in deciduous, mixed, and burned forests, and had low abundance near riparian areas

(Johnson et al. 2019). This study explained that reproductive classes of female bats exhibited a lot of variation around proximity to burned and unburned forest types, only males tended to differ in their use of burned versus unburned forest. These bats occupy a forested landscape supporting a diverse assemblage of moth prey and have small home ranges that are located closest to habitats with the greatest diversity and abundance of moths (Johnson et al. 2019). Bats in the genus *Corynorhinus*, such as the Ozark big-eared bat and Rafinesque's big-eared bat, are adapted for slow, maneuverable flight and have small home ranges near to areas with the greatest diversity and abundance of moths. Though this could make the bats susceptible to disturbances, such as fire, it seems as though prescribed fires only weakly impact the foraging selection (Johnson et al. 2019). Low-intensity fires may lead to higher abundance of big-eared bat prey (Johnson et al. 2019), and though in my study bats used burned area less than expected, about half of the tracked locations were still within burned areas.

As prescribed fire does not seem to be impacting where Ozark big-eared bats were foraging in this study, there must be other preferences that bats are selecting for when foraging. In Chapter 1, I found that there were no difference in the measured values of understory density, mature tree density, and vine and woody groundcover among areas that had been burned, thinned, burned and thinned, or not managed. There were also differences seen in other groundcover types, as well as canopy openness. In general, my results from Chapter 1 are consistent with literature that more mature tree density and vine and woody groundcover are positively associated with moth abundance, as the larvae of the moths use woody stems as a host. My study showed that the forest stand management in this area was not the only cause of differences in mature tree density and vine and woody groundcover, which means that there must be other variables that I did not include that would account for more of the variation among management areas, such as the time since management has occurred. A lot of this area has been burned, but some areas were burned so long ago that differences in forest structure may not be detected by methods used in my study.

Hot Spot Analysis

There were hot spots of Macrolepidopteran biomass identified in the home range of this Ozark big-eared bat colony, and none were identified in areas that had undergone timber thinning treatment. This could mean that Macrolepidoptera biomass could be more available for Ozark big-eared bats in areas that have not been thinned. There have been many studies examining the impacts of thinning on forest growth and yield of residual trees (Cochran and Barrett 1993), and in general, different treatments impact the residual growth of the forest differently (Dodson et al. 2011). Longevity associated with thinning depends on intensity, stand structure and composition, and site quality (Pothier 2002). Thinning allows more growing space and decreases competition for essential resources, which allows for trees to grow larger over time (Shen et al. 2019). Thinning can also create canopy gaps, which can stimulate tree seedling establishment and understory vegetation (Shen et al. 2019). My results from Chapter 1 showed that overall moth abundance (biomass) increased with more vine and woody groundcover. Mechanical thinning that would decrease woody species could have decreased moth abundance in these areas, and therefore reduced chances of these areas being labeled as hot spots. Ozark big-eared bats are moth specialists, and consume large moths, Macrolepidopterans, as a primary prey source. Preliminary evaluation of the foraging activities of these bats suggests that they prefer to forage in areas that had been treated with prescribed fire, but not areas that had undergone cutting or mechanical treatments (P. Ardapple, U. S. Fish and Wildlife Service, unpublished report), which may be due to decreased prey availability or other ecological factors in thinned areas.

In this study, it did not seem that Macrolepidoptera hot spots influenced where this colony of Ozark big-eared bats were foraging. These bats were no more likely to be close to an identified Macrolepidoptera hot spot than they were to be near a random point. This could mean that prey hot spots are not a driving force for foraging location selection and that other factors are a higher priority when bats are choosing areas to forage. Many other factors, such as distance to the roost and cover for prey could outweigh prey availability for Ozark big-eared bat foraging (P. Ardapple, U. S. Fish and Wildlife Service, unpublished report) in this area. Another reason that hot spots did not seem to be relevant to Ozark big-eared bat foraging is because of the reach of the hot spot. Using this analysis, I was not able to determine how large the hot spot was from the trap site. While the hot spot could extend a large distance from the trap site, it could potentially also be so small that it would not impact foraging area selection.

The lack of correlation between hot spots and foraging data could also be due to a small sample size. I only collected samples from 61 insect trap sites, and from these sites, only identified 6 hot spots within the bats' home range. A larger scale and more in-depth study could help in understanding what forces are driving bats to forage in certain areas and identify factors that influence Macrolepidoptera hot spots. Potential future studies using biomass data include the use of Kriging to predict areas with high values of insect abundance that could be optimal foraging locations for bats. The insect abundance measurements I found in this study may not be representative of the true insect diversity and availability in bat foraging areas due to a limited sample. Overall, my results pertaining to insect hot spots are limited and may not fully represent the forest system. Further analysis including more variables and many more insect trap sites may help to fully understand what is driving Ozark big-eared bat foraging preferences and insect hot spots.

There were a few limitations throughout this study that need to be taken into account when analyzing the implications of my data. There was a small sample size of bats studied, as these bats are endangered and it is important to minimize stress due to handling and disturbance. In addition to this, the bats that were tagged and used for this study were mostly females, as they were being retrieved from a maternity colony. This could introduce some bias as male and female bats could have different foraging patterns.

MANAGEMENT IMPLICATIONS

In general, bats in this study did not seem to be using areas that had been mechanically thinned, but this could be because of the distance of these areas from the roost. To the west of the roost, there are large burn blocks, but areas are much further to the south and east of the roost. The areas that are surrounding the Ozark big-eared bat maternity roost are areas that have been burned. This could be why a lot of the bat location data I included indicated that these areas were frequently used. Areas that had been both burned and thinned or just thinned in this area are sparse when compared to the amount of area that has been burned or not undergone management, which could also be influencing the results of my study. In general, it is beneficial to have learned from this study that bats are not actively avoiding forest areas that have been burned and thinned.

The affects that these management practices are having on the forest in the Ozark big-eared bat home range are not detrimental to this maternity colony.

Based on my results from Chapter 1, it seems that forest management does have implications on forest structure, which in turn influences insect biomass (prey availability), but I may not be seeing these results in my foraging study because of the distance of these management areas to the bat roost. I suggest management closer to the roost that will increase prey availability for these bats, such as prioritizing mature tree density and woody and herbaceous groundcover while reducing understory density and bare ground cover. My results from chapter 1 indicated that these factors were influential in increasing the amount of biomass of insects and moths that I trapped. Increased prey abundance could provide more suitable foraging habitat for these endangered bats, especially if these areas are close to the roost.

In conclusion, this study took place in an extremely complex forest system. There have been many types of management that have occurred in the area, as well as large natural disturbances and historical changes that have impacted forest structure, insect assemblages, and bat species. My study was limited with a small sample size for bats, trapping bias of insects, and the overall complexity of this forest ecosystem, but these results provide some insight on how to manage the forest for this endangered species so that they can persist in this area for years to come.

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TABLES

Table 11.1. Insect order and associated biomass (g) collected over a two-week period in June and July, 2022 in Franklin County, Arkansas.

Insect Order	Biomass (g)
Lepidoptera	751.166
Coleoptera	488.133
Megaloptera	59.644
Hemiptera	13.080
Plecoptera	12.471
Diptera	6.427
Homoptera	4.303
Odonata	3.013
Blattodea	3.001
Orthoptera	0.261
Hymenoptera	0.113
Unidentified	118.994

Table 1.12. Model selection to predict total insect biomass trapped using habitat explanatory variables.

Rank	Model	K	Δ AICc	AICc Wt	Cum. Wt	LL
1	UD[a] + VW[b] + E[c]	5	0.00	0.51	0.51	-247.62
2	NULL	2	1.89	0.20	0.71	-252.01
3	UD + VW + E + CO[d]	6	2.07	0.18	0.90	-247.42
4	UD+VW+CO+MT[e]+E	7	4.04	0.07	0.97	-247.13
5	UD+GF[f]+CO+MT+E	7	5.40	0.03	1.00	-247.18

[a] Understory density
[b] Percent vine or woody groundcover
[c] Elevation
[d] Canopy openness
[e] Mature tree density
[f] Percent grass or forb groundcover

Table 1.13. Model selection to predict Lepidopteran biomass trapped using habitat explanatory variables.

Rank	Model	K	Δ AICc	AICc Wt	Cum. Wt	LL
1	UD[a] + VW[b] + E[c]	5	0.00	0.51	0.51	-207.88
2	NULL	2	1.92	0.20	0.71	-212.28
3	UD + VW + E + CO[d]	6	2.07	0.18	0.89	-207.69
4	UD+VW+CO+MT[e]+E	7	3.98	0.07	0.96	-207.36
5	UD+GF[f]+CO+MT+E	7	5.33	0.04	1.00	-208.04

[a] Understory density
[b] Percent vine or woody grouncover
[c] Elevation
[d] Canopy openness
[e] Mature tree density
[f] Percent grass or forb groundcover

Table 1.14. Model selection to predict Macrolepidopteran biomass trapped using habitat explanatory variables.

Rank	Model	K	Δ AICc	AICc Wt	Cum. Wt	LL
1	VW[a]+CO[b]+MT[c]	5	0.00	0.52	0.52	-92.62
2	NULL	2	2.14	0.18	0.70	-97.13
3	VW+CO+MT+GF[d]	6	2.32	0.16	0.86	-92.55
4	VW+CO+MT+GF+BG[e]	7	3.74	0.08	0.94	-91.98
5	UD[f]+VW+CO+MT+E	7	4.45	0.06	1.00	-92.34

[a] Percent vine or woody groundcover
[b] Canopy openness
[c] Mature tree density
[d] Percent grass or forb groundcover
[e] Percent bare ground or dead organic groundcover
[f] Understory density

Table 1.15. Model selection to predict total insect biomass trapped using environment explanatory variables.

Rank	Model	K	Δ AICc	AICc Wt	Cum. Wt	LL
1	T[a]	3	0.00	0.64	0.64	-247.09
2	T+PMI[b]	4	1.93	0.24	0.88	-246.91
3	T+PMI+H[c]	5	4.10	0.08	0.96	-246.81
4	T+PMI+H+W[d]	6	6.52	0.02	0.99	-246.79
5	NULL	2	7.61	0.01	1.00	-252.01

[a] Temperature

[b] Percent moon illumination

[c] Humidity

[d] Windspeed

Table 1.16. Model selection to predict Lepidopteran biomass trapped using environment explanatory variables.

Rank	Model	K	Δ AICc	AICc Wt	Cum. Wt	LL
1	T[a]	3	0.00	0.65	0.65	-207.20
2	T+PMI[b]	4	2.04	0.24	0.89	-207.07
3	T+PMI+H[c]	5	4.32	0.08	0.97	-207.02
4	T+PMI+H+W[d]	6	6.71	0.02	0.99	-206.99
5	NULL	2	7.96	0.01	1.00	-212.28

[a] Temperature
[b] Percent moon illumination
[c] Humidity
[d] Windspeed

Table 1.17. Model selection to predict Macrolepidopteran biomass trapped using environment explanatory variables.

Rank	Model	K	Δ AICc	AICc Wt	Cum. Wt	LL
1	NULL	2	0.00	0.37	0.37	-97.13
2	PMI[a]	3	0.24	0.33	0.70	-96.15
3	PMI+W[b]	4	1.08	0.22	0.91	-95.42
4	PMI+W+H[c]	5	3.45	0.07	0.98	-95.42
5	PMI+W+H+T[d]	6	5.92	0.02	1.00	-95.42

[a] Percent moon illumination
[b] Windspeed
[c] Humidity
[d] Temperature

Table 1.18. Model selection to predict total insect biomass trapped using all explanatory variables.

Rank	Model Parameters	K	Δ AICc	AICc Wt	Cum. Wt	LL
1	T[a]+E[b]+W[c]+VW[d]+MT[e]	7	0.00	0.36	0.36	-241.16
2	T+E+BG[f]+GF[g]+MT+VW	8	0.24	0.32	0.68	-239.95
3	T+E+W+BG+GF+MT+VW	9	1.21	0.20	0.88	-239.06
4	T+E+W+UD[h]+BG+GF+MT+VW	10	2.29	0.12	1.00	-238.16
5	NULL	2	9.79	0	1.00	-252.01

[a] Temperature
[b] Elevation
[c] Windspeed
[d] Percent vine or woody groundcover
[e] Mature tree density
[f] Percent bare ground or dead organic groundcover
[g] Percent grass or forb groundcover
[h] Understory Density

Table 1.19. Model selection to predict Lepidopteran biomass trapped using all explanatory variables.

Rank	Model Parameters	K	ΔAICc	AICc Wt	CumWt	LL
1	T[a]+E[b]+W[c]+UD[d]+MT[e]+VW[f]	8	0.00	0.58	0.58	-199.90
2	T+E+W+UD+MT+VW+GF[g]+BG[h]	10	1.95	0.22	0.80	-198.06
3	T+E+W+UD+BG+MT	8	3.30	0.11	0.91	-201.55
4	T+E+W+UD+GF+MT	8	3.70	0.09	1.00	-201.75
5	NULL	2	10.22	0.00	1.00	-212.28

[a] Temperature
[b] Elevation
[c] Windspeed
[d] Understory Density
[e] Mature tree density
[f] Percent vine or woody groundcover
[g] Percent grass or forb groundcover
[h] Percent bare ground or dead organic groundcover

Table 1.20. Model selection to predict Macrolepidopteran biomass trapped using all explanatory variables.

Rank	Model	K	Δ AICc	AICc Wt	Cum. Wt	LL
1	Null Model	2	0.00	0.45	0.45	-97.13
2	W[a]+PMI[b]+MT[c]+VW[d]	6	1.49	0.21	0.66	-93.21
3	W+PMI+BG[e]+GF[f]+MT+VW	8	2.15	0.15	0.81	-90.93
4	W+PMI+BG+MT	6	2.82	0.11	0.92	-93,87
5	W+PMI+BG+CO[g]+GF+MT+VW	9	3.36	0.08	1.00	-90.15

[a] Windspeed
[b] Percent moon illumination
[c] Mature tree density
[d] Percent vine or woody groundcover
[e] Percent bare ground or dead organic groundcover
[f] Percent grass or forb groundcover
[g] Canopy openness

FIGURES

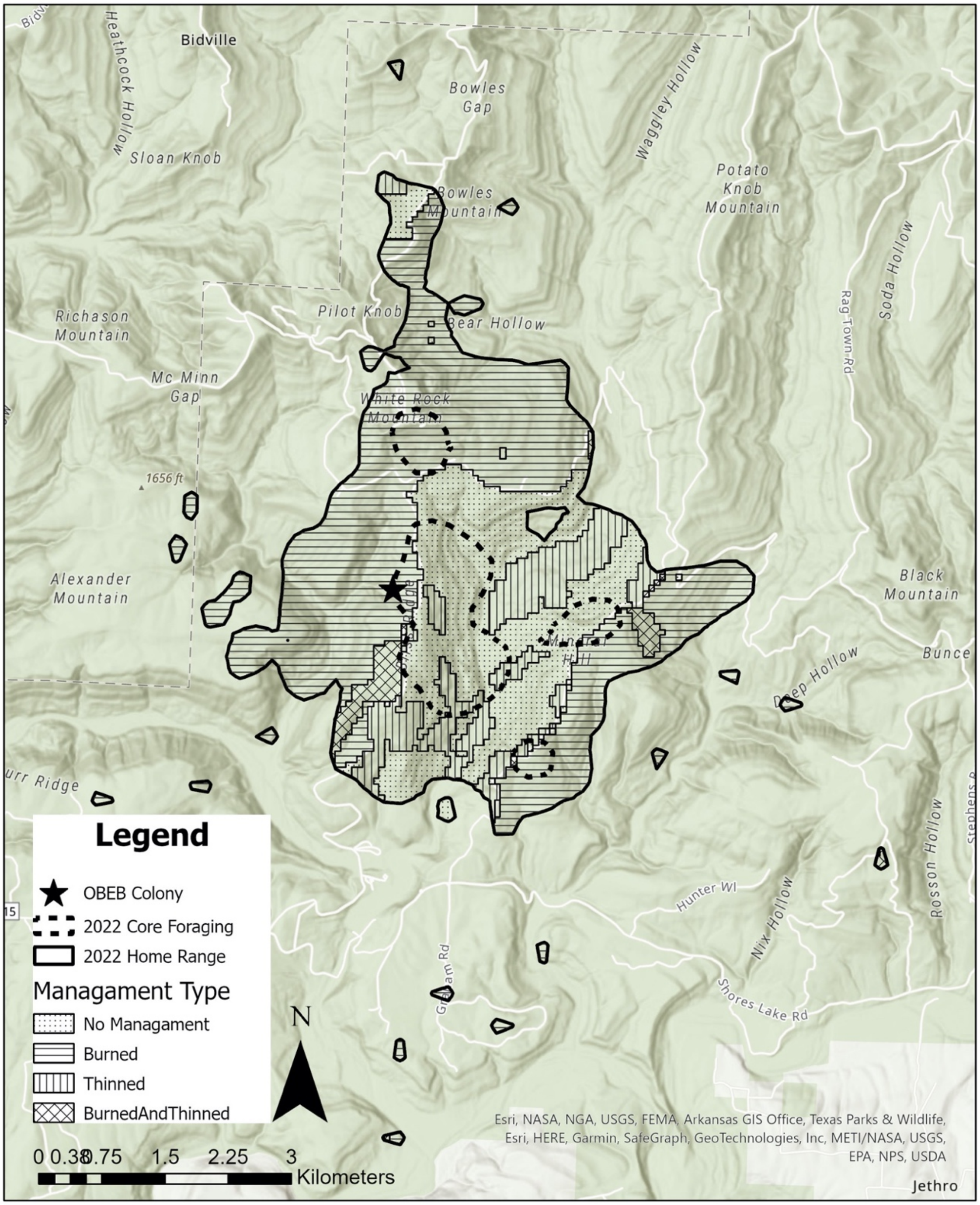


Figure 1.2. Bat home range study area including management (prescribed fire, thinned, burned and thinned, and no management), 2022 Ozark big-eared bat colony, and core foraging area.

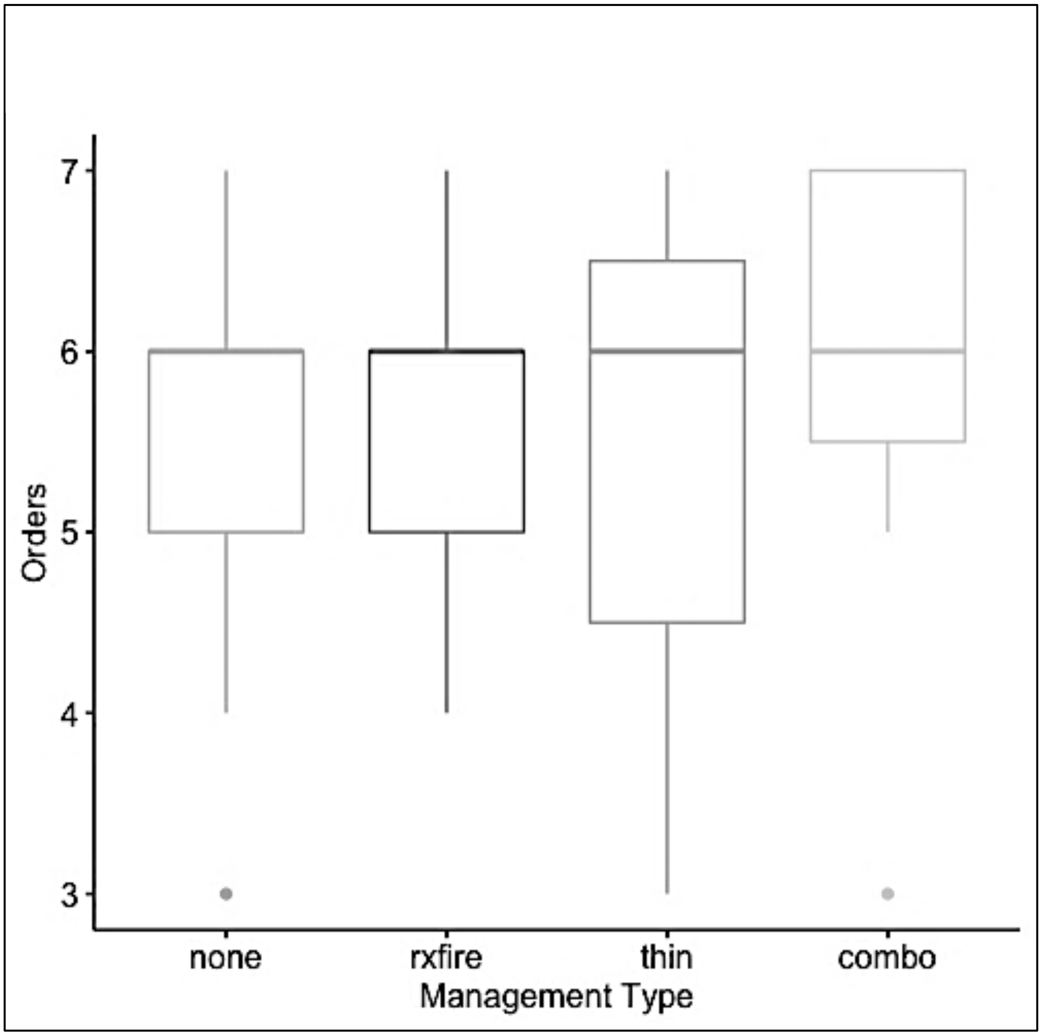


Figure 1.2. Median number of insect orders identified at a site by forest management type. Insects collected in Franklin County, Arkansas in June and July 2022.

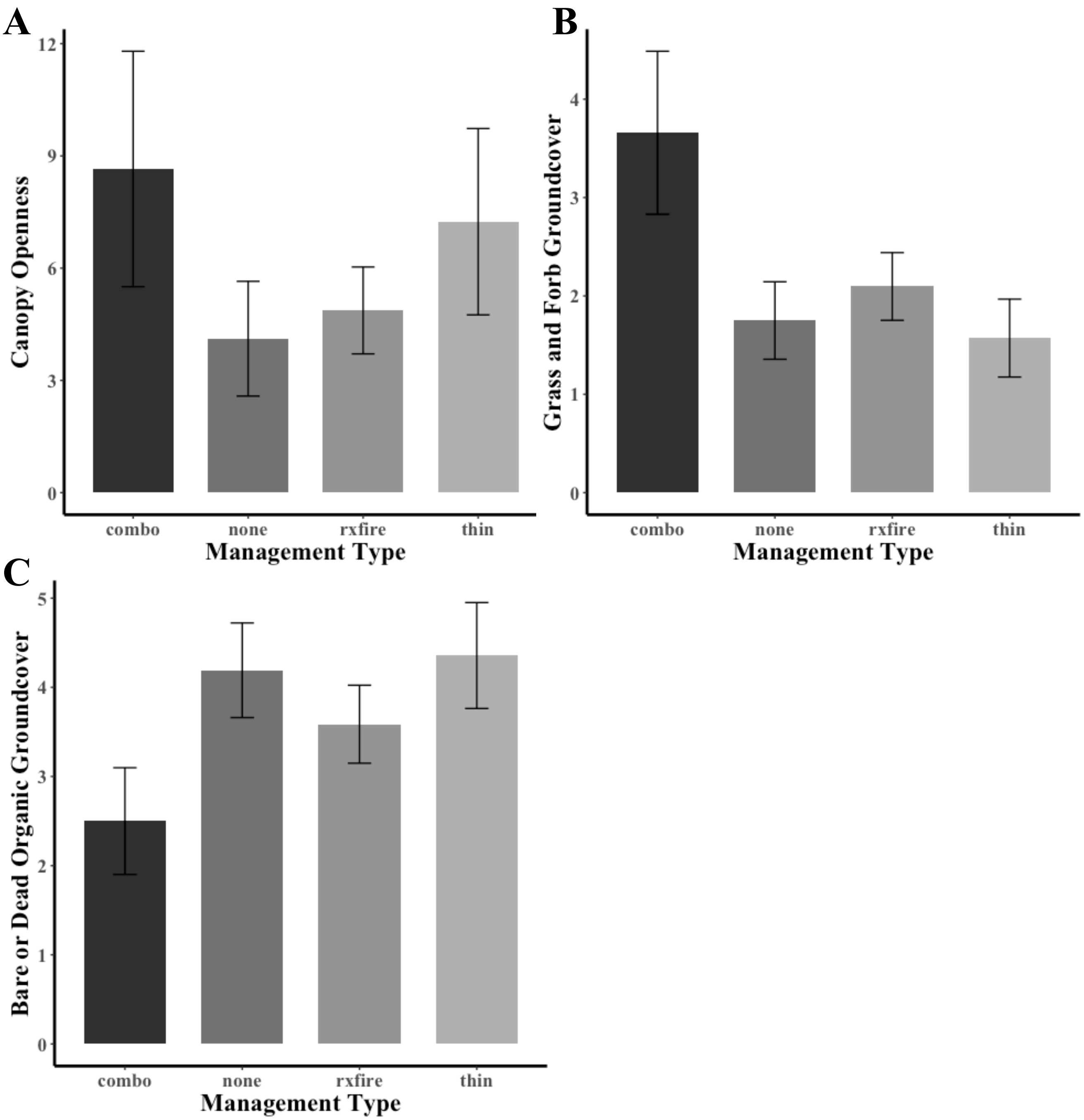


Figure 1.3. Mean value of canopy openness among forest management types (A), mean score of percent grass or forb groundcover among forest management types (B), and mean score of percent bare or organic groundcover among forest management types (C). Measurements taken in Franklin County, Arkansas during June and July 2022.

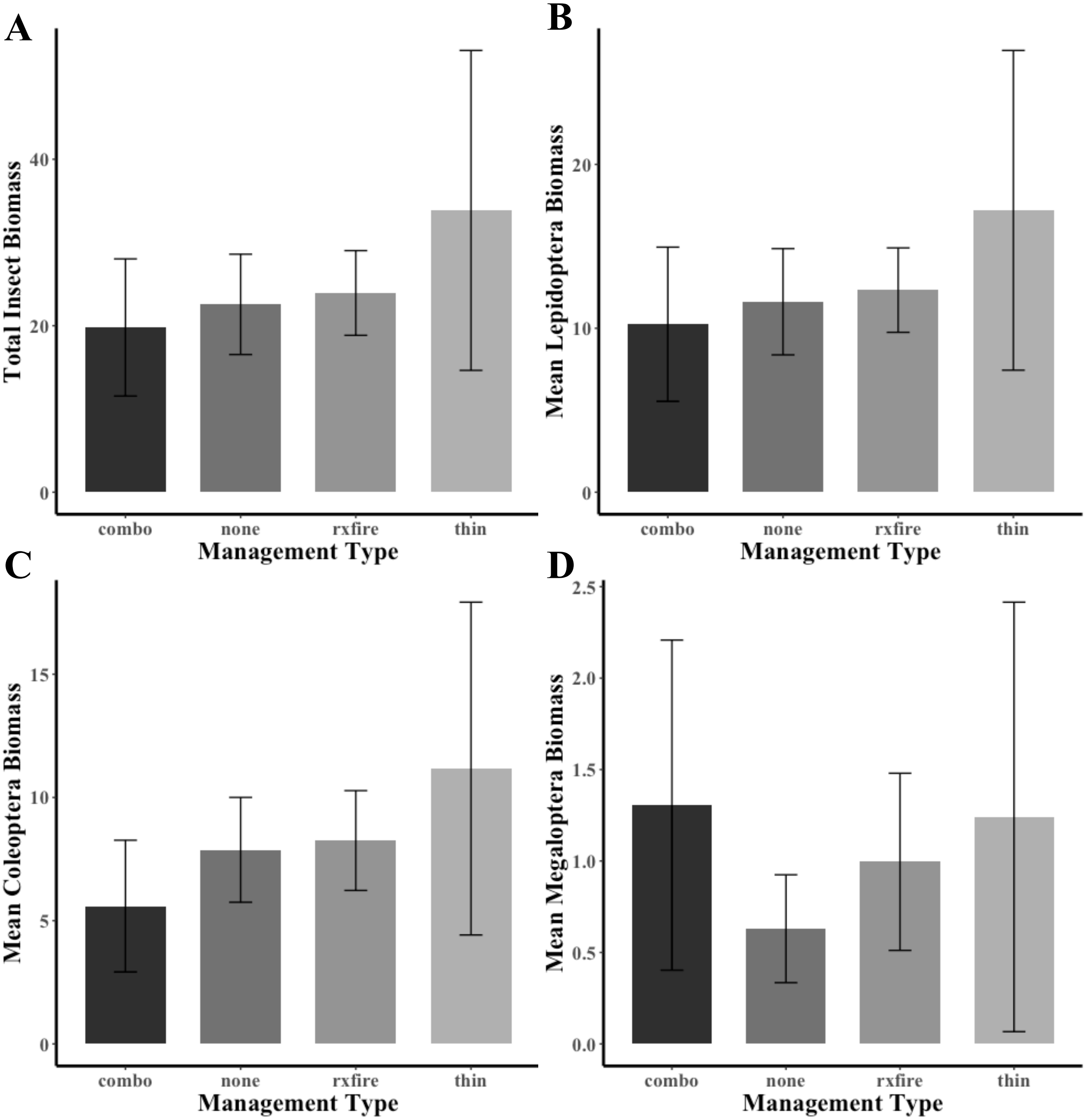


Figure 1.4. Mean value of total insect biomass trapped among forest management types (A), mean value of Lepidoptera biomass trapped among forest management types (B), mean value of Coleoptera biomass trapped among forest management types (C), and mean value of Megaloptera biomass trapped among forest management types. Insects trapped in June and July 2022 in Franklin County, Arkansas.

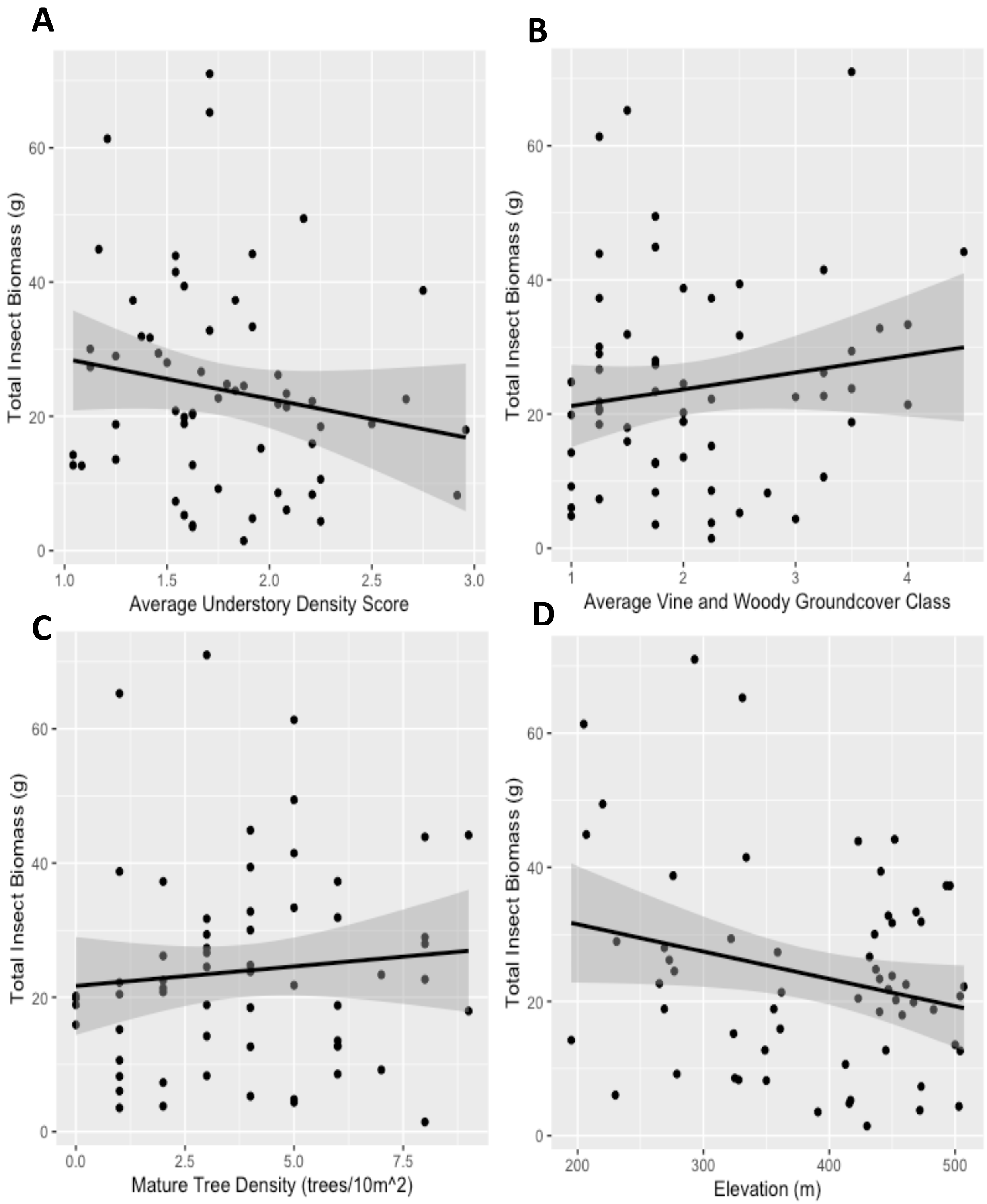


Figure 1.5. Predicted values (±95% CI) for average understory density (A), percent vine and woody groundcover (B), mature tree density (C) and elevation (D) in the competitive models for total insect biomass trapped. Insects collected in June and July 2022 in Franklin County, Arkansas.

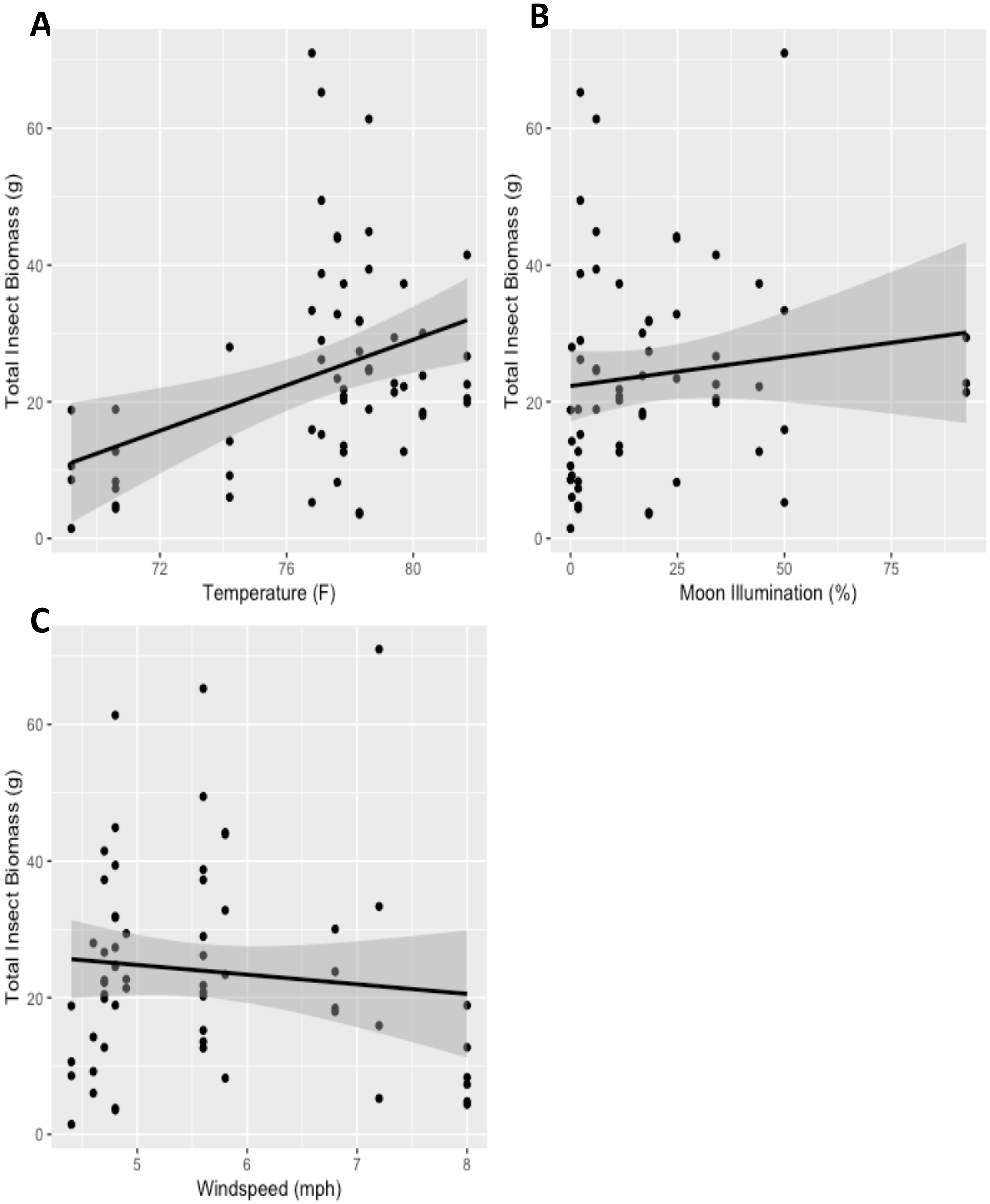


Figure 1.6. Predicted values (±95% CI) for temperature (A), percent moon illumination (B) and windspeed (C) in competitive models for predicting total insect biomass trapped. Insects trapped in June and July 2022 in Franklin County, Arkansas.

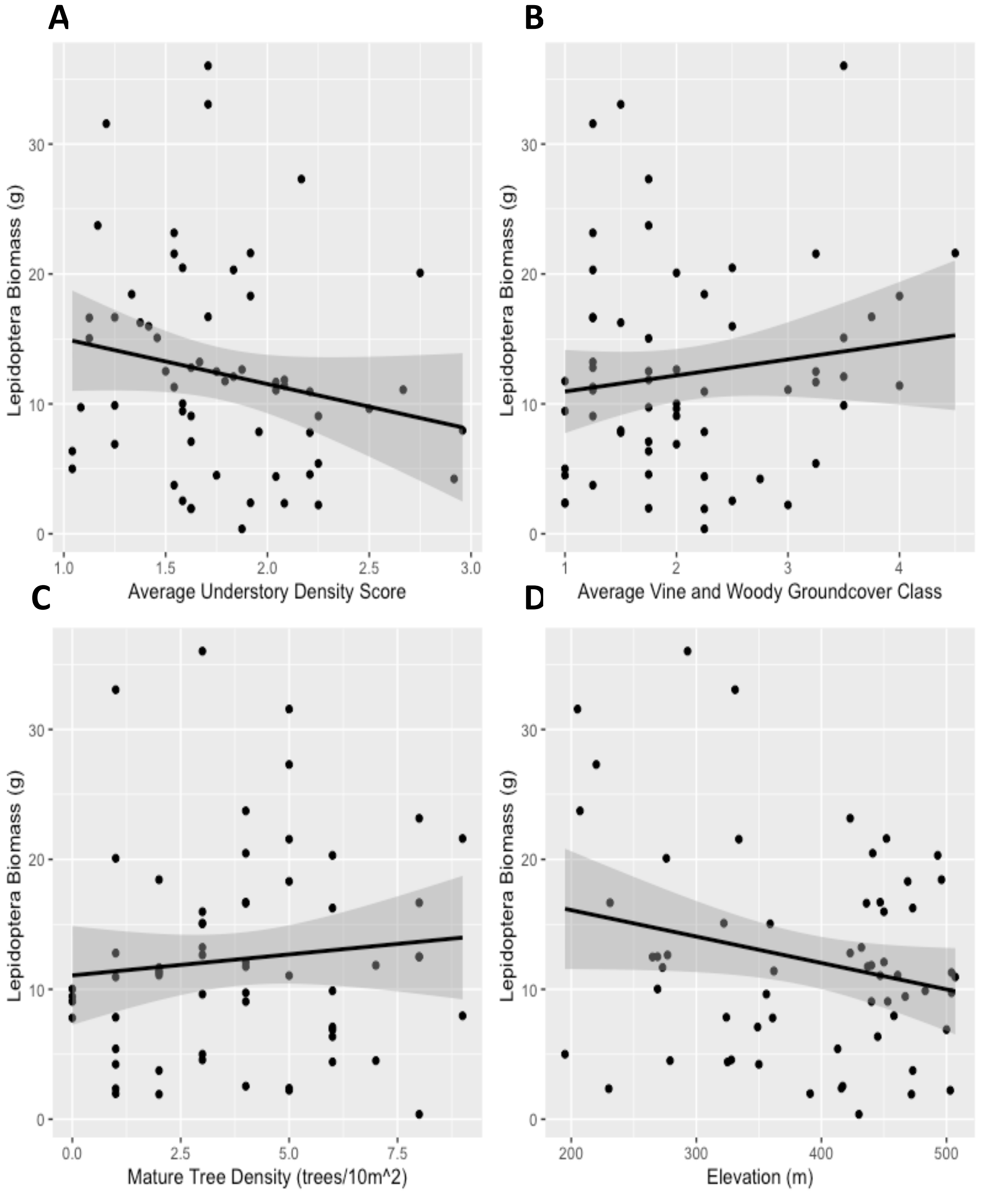


Figure 1.7. Predicted values (±95% CI) for average understory density (A), percent vine and woody groundcover (B), mature tree density (C) and elevation (D) in competitive models for predicting Lepidoptera biomass trapped. Insects trapped in June and July 2022 in Franklin County, Arkansas.

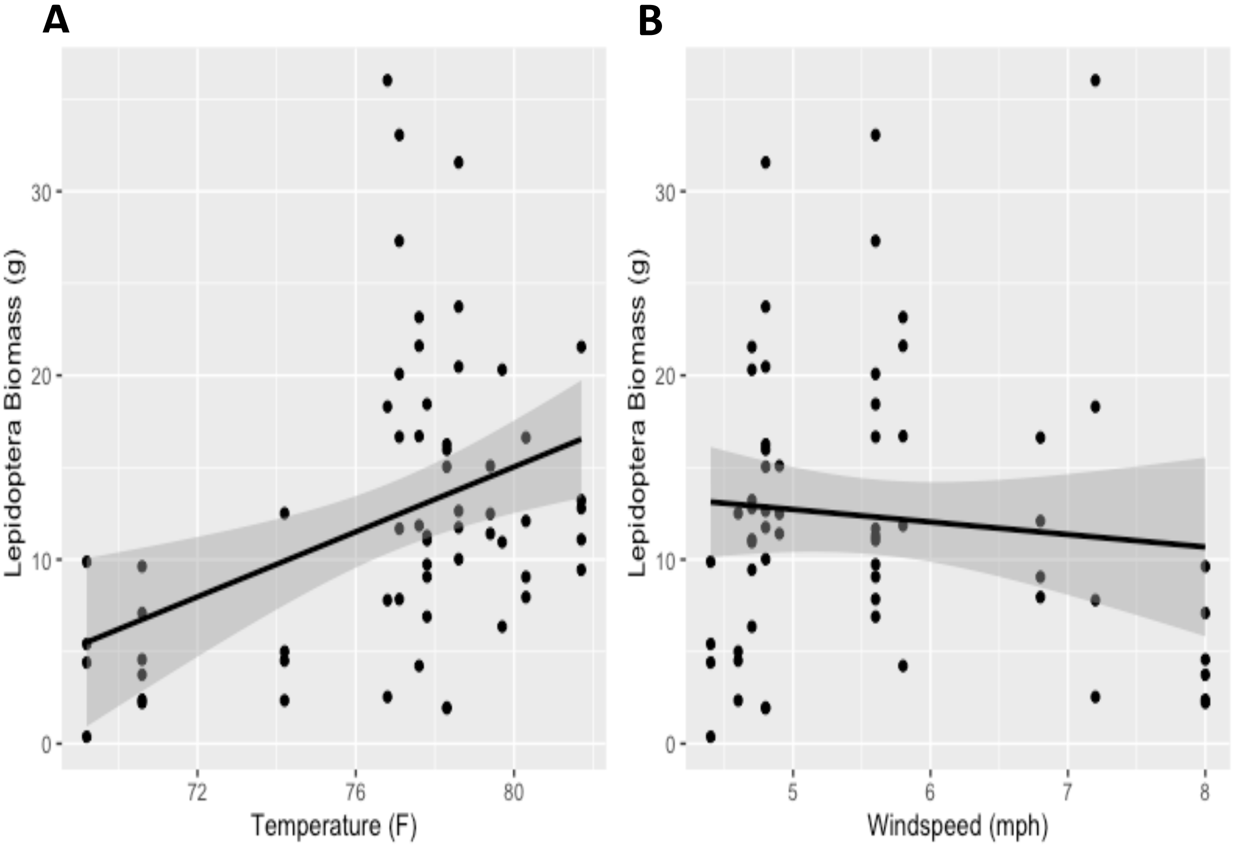


Figure 1.8. Predicted values (±95% CI) for temperature (A) and windspeed (B) in competitive models for predicting Lepidoptera biomass trapped. Insects trapped in June and July 2022 in Franklin County, Arkansas.

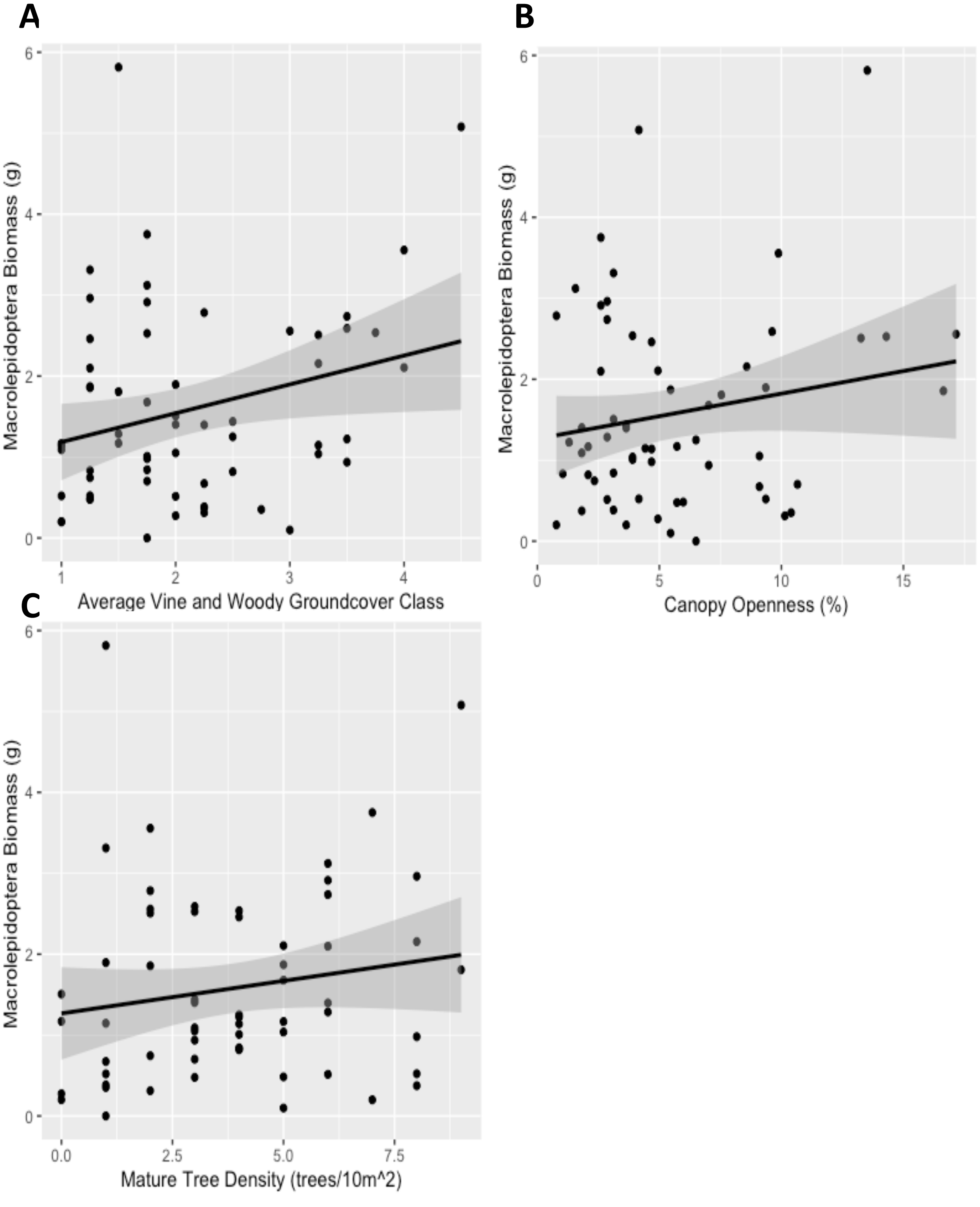


Figure 1.9. Predicted values (±95% CI) for percent vine and woody groundcover (A), canopy openness (B) and mature tree density (C) in competitive models for predicting Macrolepidoptera biomass trapped. Insects trapped in June and July 2022 in Franklin County, Arkansas.

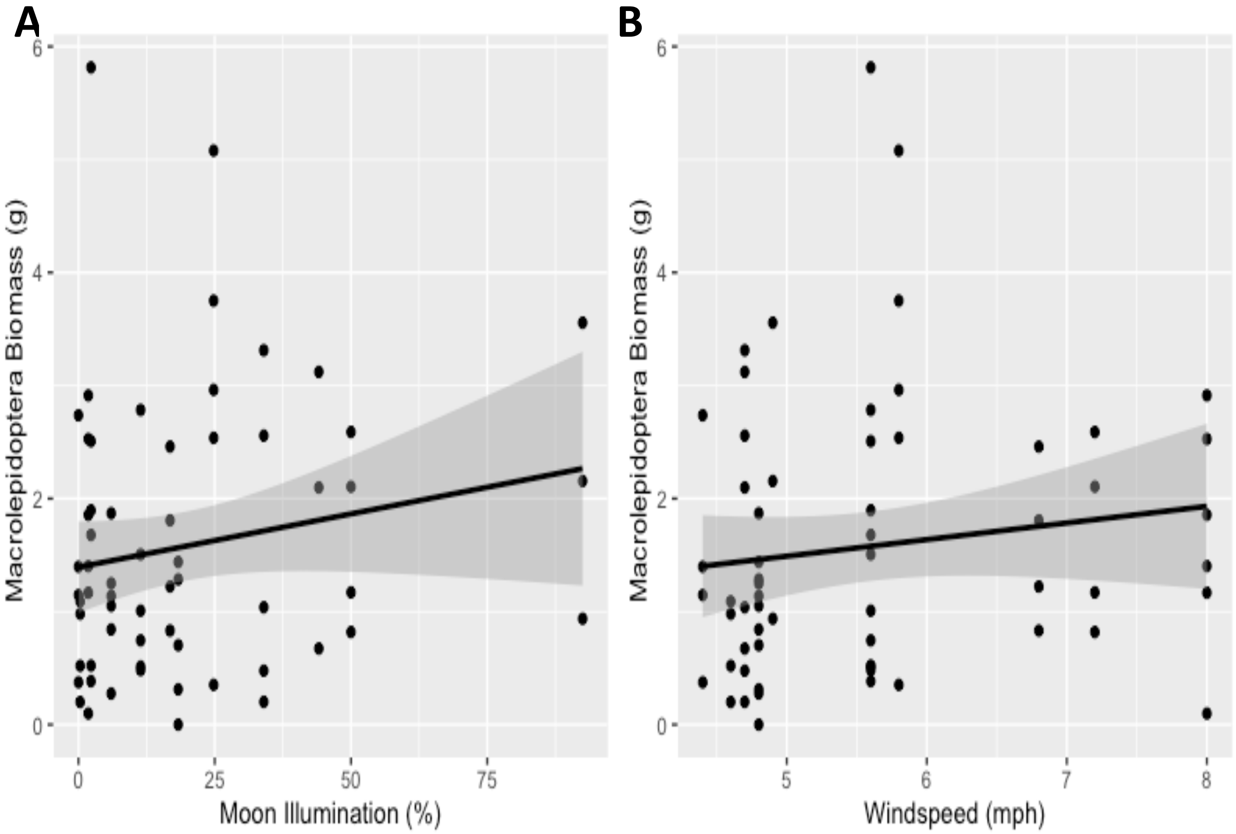
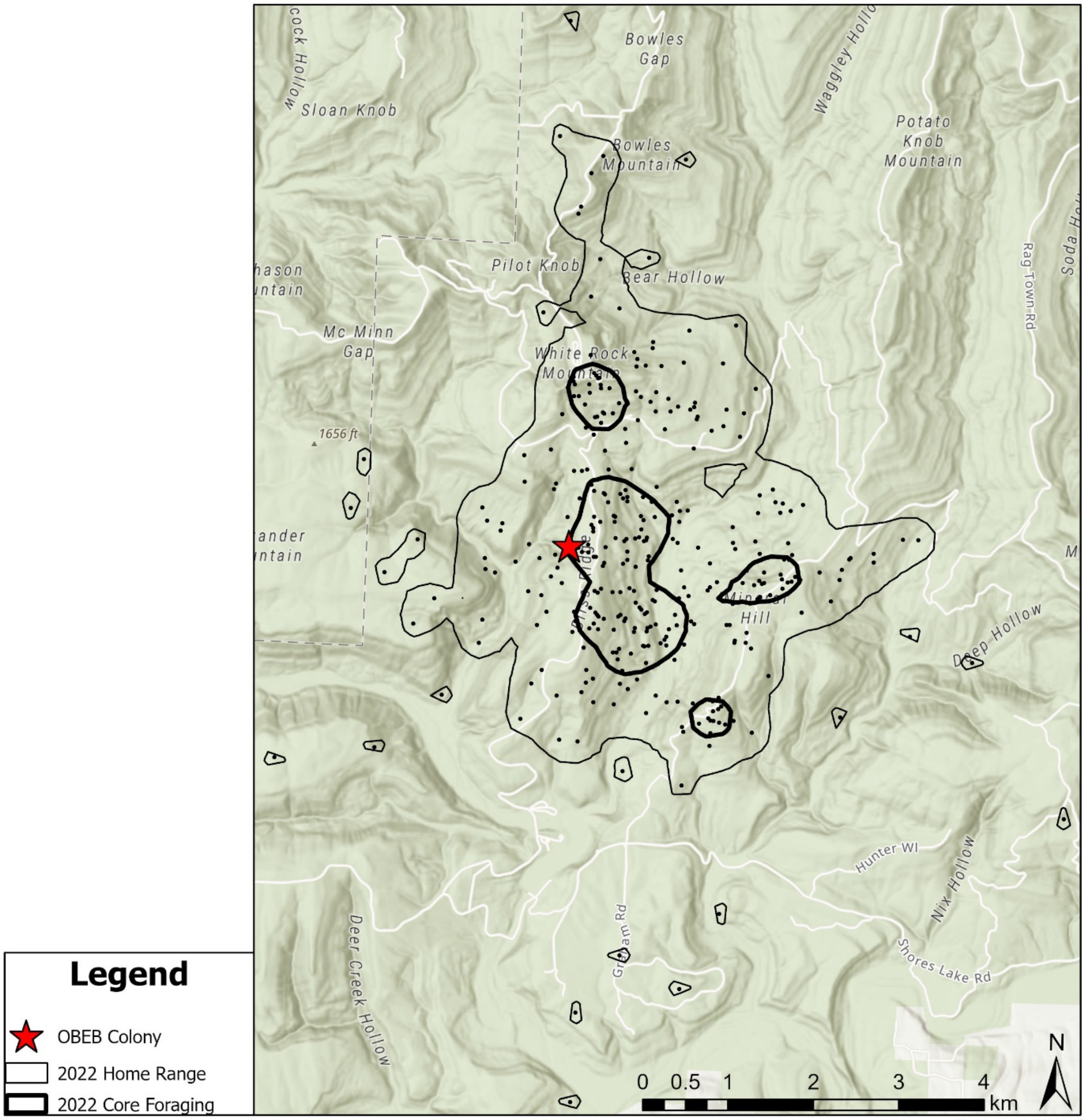
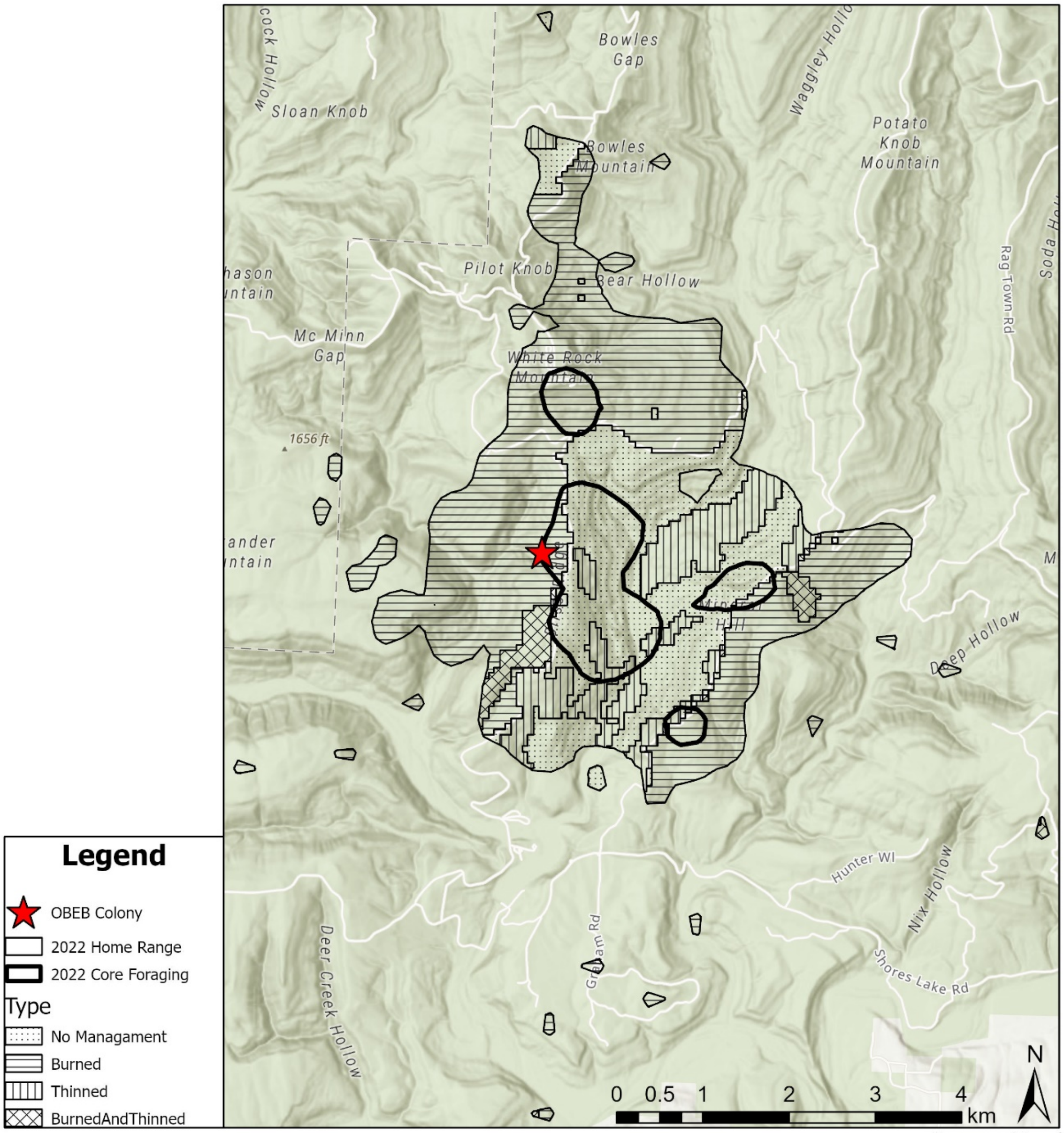


Figure 1.10. Predicted values (±95% CI) for percent moon illumination (A) and windspeed (B) in competitive models to predict Macrolepidoptera biomass trapped. Insects trapped in June and July 2022 in Franklin County, Arkansas.



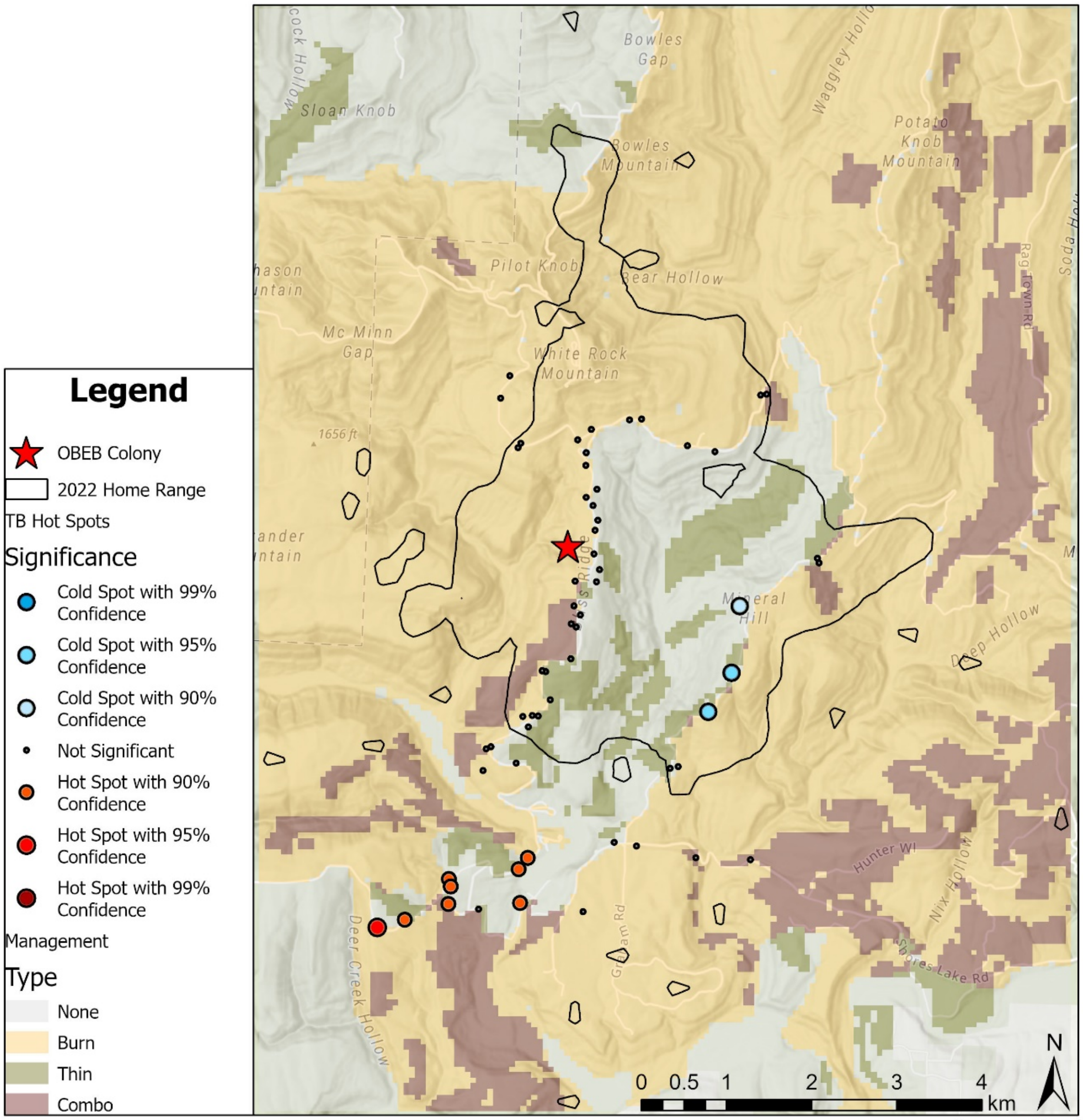
Gemma Marsh
June 2023

Figure 2.1. Ozark big-eared bat colony 2022 home range, core foraging areas, and radio-tracked bat locations (black points). Data taken in June and July 2022, in Franklin County, Arkansas.



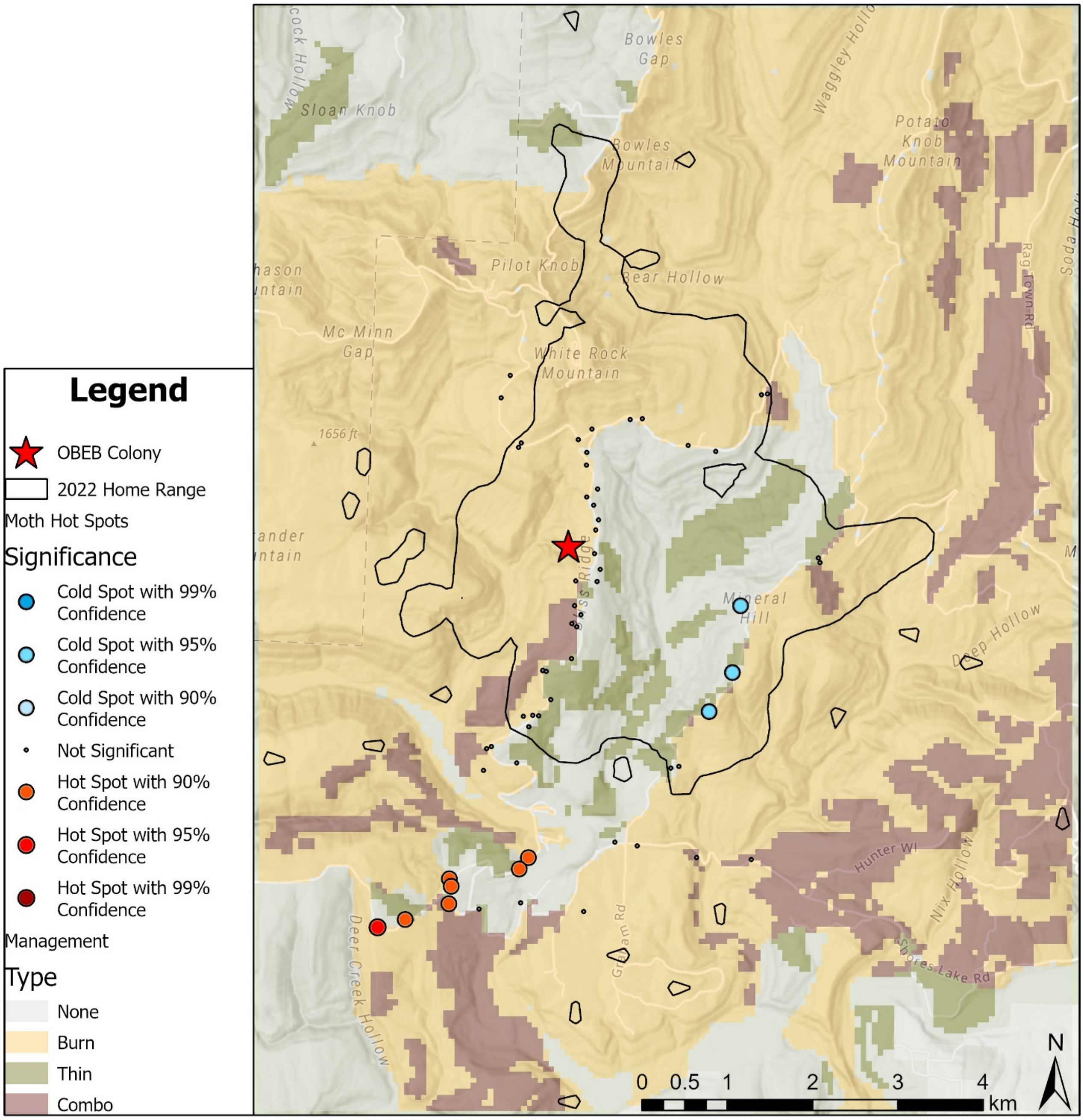
Gemma Marsh
June 2023

Figure 2.2. Ozark big-eared bat colony 2022 home range and core foraging areas. Forest management types denoted by shading. Ozark big-eared bat colony denoted by red star. Ozark big-eared bat colony and home range located in Franklin County, Arkansas.



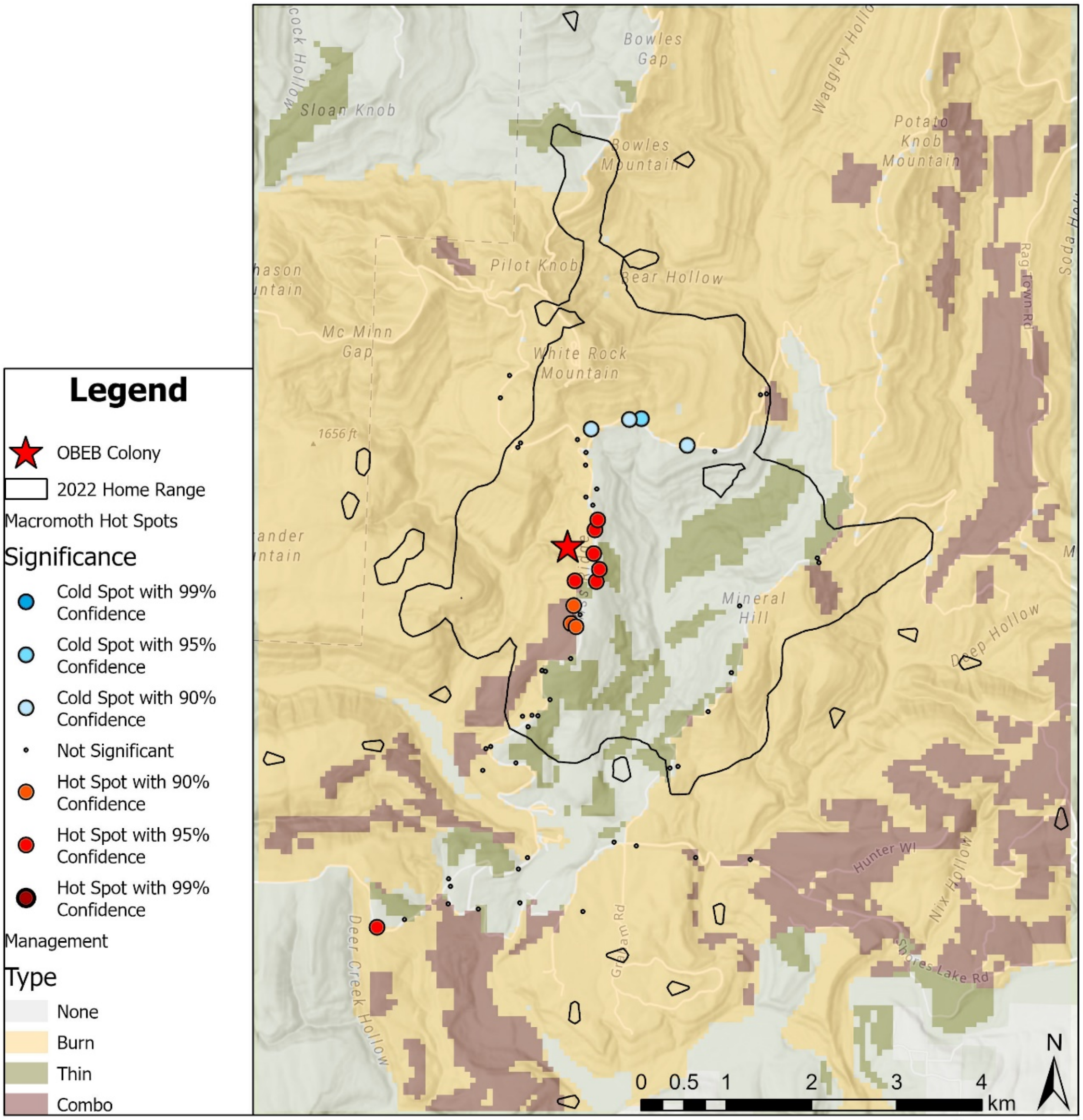
Gemma Marsh
June 2023

Figure 2.3. Hot spots of total insect biomass trapped near Ozark big-eared bat colony. One hot spot with 95% confidence was determined (red point), and it was not located within the 2022 home range. Forest management types shaded throughout region. Ozark big-eared bat colony and home range located in Franklin County, Arkansas.



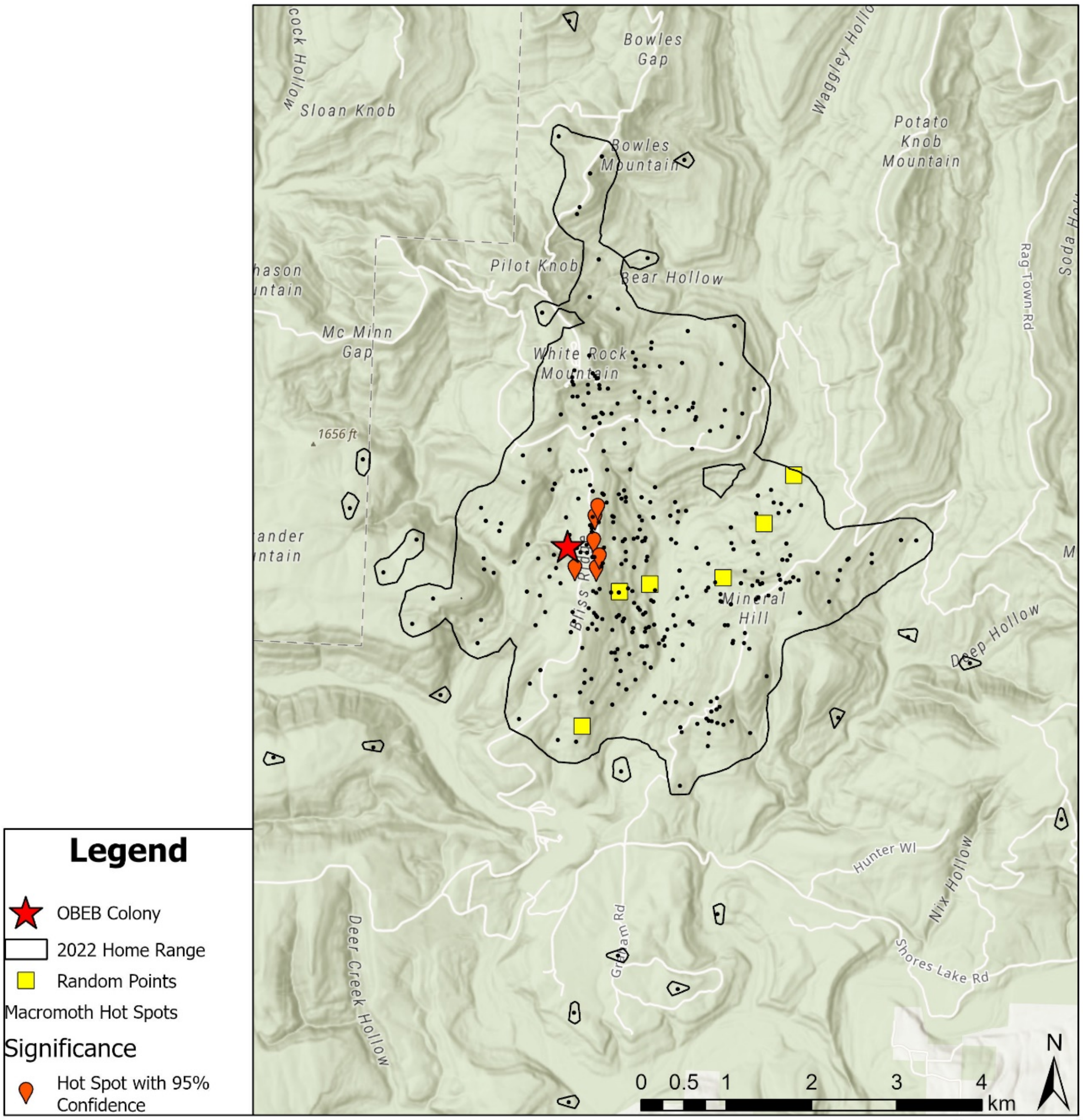
Gemma Marsh
June 2023

Figure 2.4. Hot spots of lepidoptera biomass trapped near Ozark big-eared bat colony. Similar to the total biomass analysis, one hot spot with 95% confidence was determined (red point), and it was outside the home range of the colony. Forest management types shaded throughout region. Ozark big-eared bat colony and home range located in Franklin County, Arkansas.



Gemma Marsh
June 2023

Figure 2.5. Hot spots of Macrolepidoptera biomass trapped near Ozark big-eared bat colony (red star). Six hot spots (red points) with 95% confidence located within the 2022 home range of the bat colony. Forest management shaded throughout region. Ozark big-eared bat colony and home range located in Franklin County, Arkansas.



Gemma Marsh
June 2023

Figure 2.6. Six Macrolepidoptera hot spots (orange) located within the 2022 home range of the Ozark big-eared bat colony and six randomly determined locations (yellow squares) within the 2022 home range. All 2022 bat locations denoted by black points. Ozark big-eared bat colony and home range located in Franklin County, Arkansas.